

Western Yellow-Billed Cuckoo (*Coccyzus americanus occidentalis*)

Legal Status

State: Endangered

Federal: Threatened, Bureau of Land Management Sensitive, U.S.

Fish and Wildlife Service (USFWS) Bird of Conservation Concern, U.S. Forest Service Sensitive

Critical Habitat: 79 FR 67154-67155

Recovery Planning: N/A

Notes: In 2014, the USFWS published a final rule describing the determination of threatened status for the western distinct population segment (DPS) of yellow-billed cuckoo (*Coccyzus americanus*) (i.e., western yellow-billed cuckoo) (79 FR 59992–60038).



Courtesy of Murrelet Halterman, PhD.

Taxonomy

Two subspecies of the yellow-billed cuckoo are recognized—western yellow-billed cuckoo (*C. a. occidentalis*) and eastern yellow-billed cuckoo (*C. a. americanus*)—although the validity of the taxonomic grouping has been debated based on morphometric measurements (e.g., wing length) (Banks 1988, 1990; Franzreb and Laymon 1993). Banks (1988) initially found statistically insignificant differences in wing length, bill length, and upper mandible depth between alleged subspecies. Revised analyses were performed given statistical and methodological errors in the Banks (1988) study. The updated Banks (1990) study found significant differences in wing and bill size between eastern and western cuckoos, but it still concluded that the subspecies should not be recognized. Franzreb and Laymon (1993) used Banks's data and determined that there were significant differences between eastern and western cuckoos in wing, tail, and bill lengths, as well as bill depth, in addition to potential behavioral, vocal, and ecological differences. Franzreb and Laymon (1993) concluded that recognition of the two subspecies should be retained until further examination determined otherwise. The two subspecies are separated by

geographic distribution, with the boundary between the two subspecies considered to be the Pecos River in Texas (Hughes 1999). It should be noted that the USFWS refers to the western U.S. DPS in the October 2011 annual review (76 FR 66370-66439) rather than the state-listed western yellow-billed cuckoo subspecies.

Descriptions of the species' physical characteristics can be found in Hughes (1999).

Distribution

General

The western yellow-billed cuckoo's historical geographic range is southwestern British Columbia, western Washington, northern Utah, central Colorado, western Texas, south and west to California, and southern Baja California, Sinaloa, and Chihuahua in Mexico (Hughes 1999) (Figure SP-B13). The western yellow-billed cuckoo is rare and local in the southwestern United States. It breeds along the major river valleys in southern and western New Mexico, and central and southern Arizona. In California, the western yellow-billed cuckoo's breeding distribution is now thought to be restricted to isolated sites in the Sacramento, Amargosa, Kern, Santa Ana, and Colorado River valleys (Laymon and Halterman 1987). During surveys in 1999 and 2000 western yellow-billed cuckoos were not found on the Amargosa and Santa Ana rivers (Laymon, pers. comm. 2012).

Distribution and Occurrences within the Plan Area

Historical

The California Natural Diversity Database (CNDDB) contains 28 historical (i.e., pre-1990) occurrence records dating from 1917 to 1986. Of the known occurrences, 24 are from 2 years: 1977 (13) and 1986 (11). Single known occurrences are from 1917, 1945, 1978, and 1983. Of the historical known occurrences in the Plan Area, 23 are from the Lower Colorado River, with 14 known occurrences from Imperial County, ranging the Palo Verde area to the U.S.-Mexico border; 6 from eastern Riverside County in the Blythe area; and 2 from San Bernardino County in the Needles area. Five of the historical known occurrences are from the Amargosa River, Tecopa, China Ranch, and Independence

areas in Inyo County, and 2 are from the Mojave River in the Upper Narrows and Hodge areas in San Bernardino County. Of 28 historical known occurrences, the majority are on public land.

Recent

In the Sacramento Valley, the south coast (including Ventura and Los Angeles counties), and Kern County, yellow-billed cuckoos were considered common to numerous in late the 1800s, but only fairly common by 1920s (Hughes 1999). By the 1950s, the subspecies had been extirpated north of Sacramento Valley (Hughes 1999). The species may also no longer breed in the Amargosa and Santa Ana rivers (Laymon, pers. comm. 2012).

The CNDDB contains nine recent (i.e., since 1990) occurrences for the Plan Area: a 1991 known occurrence in the Alabama Hills near Lone Pine, a 1998 known occurrence from the Laguna Dam area of the Colorado River in Imperial County, a 2009 occurrence north of the Cibola National Wildlife Refuge (NWR), a 2009 occurrence in the Imperial NWR area, and three 2009 occurrences along the Colorado River in the Palo Verde Ecological Reserve in Riverside County (Figure SP-B13) (CDFW 2013).

Natural History

Habitat Requirements

This discussion is limited to breeding habitat requirements for western yellow-billed cuckoo in California. Breeding habitat primarily consists of large blocks, or contiguous areas, of riparian habitat, particularly cottonwood–willow riparian woodlands (66 FR 38611–38626) (see Table 1). From a survey conducted from northern Kern and Inyo counties south in 1986 and from southern Kern and Mono counties north in 1987, Laymon and Halterman (1989) proposed that optimum habitat patches for the western yellow-billed cuckoo are greater than 200 acres in size and wider than 1,950 feet; sites 101 to 200 acres in size and wider than 650 feet were suitable; sites 50 to 100 acres in size and 325 to 650 feet were marginal; and sites smaller than these dimensions were unsuitable. Western yellow-billed cuckoo prefers dense riparian thickets with dense low-level foliage near slow-moving water sources. Nests are constructed in willows on

BIRDS**Western Yellow-Billed Cuckoo (*Coccyzus americanus occidentalis*)**

horizontal branches in trees, shrubs, and vines, but cottonwoods (*Populus* spp.) are used extensively for foraging, and humid lowland forests are used during migration (Hughes 1999). Of 95 detected nests at the South Fork Kern River, all were in willows, with one exception in a cottonwood (Laymon 1998). Along the Santa Ana River, 92% of nests were in willows, with one nest in a mistletoe clump in a cottonwood and one in an alder (*Alnus* spp.) (Laymon 1998). Nests along the Sacramento River have been found in willow, cottonwood, and alder, and also, although rarely, in orchards (Laymon 1998).

Laymon (1998) presents some detailed habitat information for the Bill Williams River in the Lake Havasu area in Arizona. This area is the most relevant to the Plan Area populations in the lower Colorado River area. Of 14 nests detected in the Bill Williams River, 11 were in willows, 1 in a cottonwood, and 2 in tamarisk (*Tamarix* spp.). Canopy closure averaged 77% and range from 51% to 92%; shrub averaged 33% with a range of 5% to 85%. The average distance of nests to water was 135 feet with a range of 0 to 575 feet.

Table 1. Habitat Associations for Western Yellow-Billed Cuckoo

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Riparian woodland and forest	Nesting and foraging	Primary	Patch size > 198 acres; width > 1,270 feet; dense vegetation	Laymon and Halterman 1989

Foraging Requirements

Yellow-billed cuckoos generally forage for lepidopteran larvae (caterpillars) and other large insects such as katydids by gleaning (Hughes 1999; Laymon 1998). They will also occasionally prey on small lizards, frogs, eggs, and young birds (Gaines 1999; Laymon 1998). Foraging occurs extensively in cottonwood riparian habitat (Hughes 1999).

Reproduction

In the western United States, nests are typically constructed in willows (*Salix* spp.), Fremont cottonwood (*Populus fremontii*),

mesquite (*Prosopis* spp.), hackberry (*Celtis* spp.), soapberry (*Sapindus saponaria*), alder (*Alnus* spp.), or cultivated fruit trees on horizontal branches or vertical forks of the large tree or shrub (Hughes 1999). Nests are generally placed between 1 and 6 meters (3 and 20 feet) above the ground and concealed by foliage, especially from above (Hughes 1999). Nest sites in arid regions are restricted to relatively humid river bottoms, ponds, swampy areas, and damp thickets (Hughes 1999). Both the male and female build the nest from twigs (approximately 15 centimeters [6 inches] long) likely collected within 10 meters (33 feet) of the nest site (Hughes 1999).

The western yellow-billed cuckoo has a short breeding season, lasting only about 4 months from time of arrival on breeding grounds in the spring to fall migration (see Table 2). Western yellow-billed cuckoos typically lay a single clutch per season of two or three eggs (average is just over two eggs, and up to four eggs per clutch is known) in mid-June to mid-July, and incubation occurs over 9 to 11 days (Hughes 1999; Johnson et al. 2008). However, Laymon (1998) reports that in years of abundant resources, double- and even triple-clutching in a season can occur along the South Fork Kern River; over a 12-year period, double-clutching occurred less than half of the study years, and triple-clutching only occurred one year. Double-clutching has not been observed at the Bill Williams River site near the Colorado River (Laymon 1998). Development of the young is very rapid, with fledging occurring in 6 to 9 days; the entire breeding cycle may be only 17 days from egg laying to fledging of the young (Hughes 1999). Fledglings are dependent upon parents for up to 3 weeks following fledging (Johnson et al. 2008). Females often switch mates between broods within years and usually select a new mate in subsequent years. They can also be communal nesters with 2 females laying eggs in a nest and tending the young. Nests often have a helper male that tends the young (Laymon, pers. comm. 2012). The yellow-billed cuckoo has been noted to be both an intraspecific and interspecific brood parasite (Hughes 1999); however, this appears to only occur in the eastern yellow-billed cuckoo. The western yellow-billed cuckoo apparently is rarely parasitized by the brown-headed cowbird (*Molothrus ater*), possibly because its short breeding period reduces the chance of successful nest parasitism (Hughes 1999).

Table 2. Key Seasonal Periods for Western Yellow-Billed Cuckoo

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding					X	X	X	X	X			
Migration									X	X		

Notes: Breeding in late May is rare.

Sources: Laymon 1998; Hughes 1999; Gaines 1999.

Spatial Behavior

Spatial behavior patterns in the western yellow-billed cuckoo include migration, territory use, and dispersal from natal sites, as summarized in Table 3.

The western yellow-billed cuckoo is a long-distance migrant, although details of its migration patterns are not well known (Hughes 1999). It is a relatively late spring migrant, arriving on the breeding grounds starting mid- to late May, but more commonly in June, and leaving from late August to early September (Franzreb and Laymon 1993; Gaines 1999) (Table 2). The migratory route of the western yellow-billed cuckoo is not well known because few specimens collected on wintering grounds have been ascribed to the western or eastern subspecies. The western yellow-billed cuckoo likely moves down the Pacific Slope of Mexico and Central America to northwestern South America (Hughes 1999).

Western yellow-billed cuckoos may have variable breeding territory sizes, with territories reported to be as small as 10 acres on the Colorado River (Laymon and Halterman 1989), but with a range of 20 to 100 acres on the South Fork Kern River (Laymon 1998). Recent data from radio telemetry studies on the Colorado, San Pedro, and Rio Grande rivers have shown larger home ranges. Cuckoos on the Rio Grande in New Mexico used an average of 204 acres (Sechrist et al. 2009), while cuckoos on the San Pedro River in Arizona, averaged about 125 acres (Halterman 2009). On the Colorado River in Arizona and California, cuckoos home ranges averaged about 95 acres (McNeil et al. 2010; McNeil et al. 2011a, 2011b). Whether western yellow-billed cuckoos are “territorial” in the sense of defending a spatially

defined area is uncertain, although individuals have been observed to aggressively supplant each other (Hughes 1999).

Dispersal and the degree to which the western yellow-billed cuckoo shows site fidelity is largely unknown. The absence of pairs on known breeding sites in some years and presence of breeding birds on previously vacant sites demonstrates that breeding may not occur in the same location every year (Gaines and Laymon 1984). However, some breeding pairs along the South Fork Kern River have returned to the same nest territories for up to 3 years (unpublished data reported by Laymon 1998). Limited banding data indicate birds returning to breeding sites within 1.2 miles of natal sites (Hughes 1999), but too few birds have been banded and monitored to document typical dispersal patterns with any confidence. Along the South Fork Kern River, all banded individuals that have been resighted in the same area have been males (Laymon 1998).

Table 3. Spatial Behavior by Western Yellow-Billed Cuckoo

Type	Distance/Area	Location of Study	Supporting Information
Home Range (Territory?)	As small as 10 acres	Colorado River	Laymon and Halterman 1989
Home Range	20–100 acres	South Kern River	Laymon 1998

Ecological Relationships

Intraspecific and interspecific and community relationships are not well understood for the western yellow-billed cuckoo. The eastern yellow-billed cuckoo is an intraspecific and interspecific brood parasite, but this behavior has not been documented in the western yellow-billed cuckoo (Hughes 1999). Where brood parasitism does occur, yellow-billed cuckoos may be mobbed and harassed by other native birds such as American robin (*Turdus migratorius*) (Hughes 1999). Otherwise, there is no information regarding intraspecific and interspecific relationships or competition (Hughes 1999).

Western yellow-billed cuckoos are vulnerable to predation by other birds, particularly by raptors during migration, snakes, and mammals (Hughes 1999). Laymon (1998) reports that red-

shouldered hawk (*Buteo lineatus*) and northern harrier (*Circus cyaneus*) have preyed on nestlings and that cuckoos chase western scrub-jay (*Aphelocoma californica*) and loggerhead shrike (*Lanius ludovicianus*) away from nests.

Presence and successful breeding by yellow-billed cuckoos may be limited by available resources. At occupied breeding sites, nesting success may be limited by available food sources. Cuckoo chicks hatch asynchronously, so the nest may contain unhatched eggs and young of various ages (Hughes 1999). The youngest chick in a brood may not be fed when food sources are in short supply, and birds may not reproduce at all when insufficient food is available (Hughes 1999). It also appears that increased food availability has a positive effect on clutch size (Martin 1987; Laymon 1998). A study of the effects of climate on yellow-billed cuckoo found that nesting by eastern yellow-billed cuckoos in the 2003 and 2004 breeding seasons only occurred at sites where caterpillars were more abundant (Anders and Post 2006) (also see discussion below on climate effects). Laymon (1998) reports that western yellow-billed cuckoos may produce multiple clutches along the South Fork Kern River when food sources are abundant.

Population Status and Trends

Global: Declining (NatureServe 2010)

State: Declining (Laymon 1998)

Within Plan Area: Same as above

Western yellow-billed cuckoo was once considered common to numerous in the Sacramento Valley, along the southern coast of California from Ventura to Los Angeles counties, and in Kern County in the late 1800s, but it was considered only fairly common by the 1920s (Gaines 1974; Gaines and Laymon 1984). The numbers of yellow-billed cuckoos in California and other western areas had declined markedly into the 1980s with loss of riparian habitats (Laymon and Halterman 1987). Surveys in 1986 and 1987 showed a decline from 123 to 163 pairs in 1977 to 30 to 33 pairs in 1987, or a 73% to 82% decline over this 10-year period (Laymon 1998). The most recent statewide surveys in 1999 and 2000, including the Sacramento, Kern, and Lower Colorado rivers (1999 only), as well as other areas with smaller amounts of habitat, documented 41 to 45

pairs and 49 unmated birds in 1999, and 61 to 67 pairs and 61 to 68 unmated birds in 2000 on the Sacramento and Kern rivers (Halterman et al. 2003). Although the number of detected pairs was higher in 1999-2000 compared to 1986-1987, there were still substantially fewer pairs than detected in 1977.

The western yellow-billed cuckoo suffered substantial range reductions in the twentieth century due to loss of riparian habitat (Laymon and Halterman 1987). The species was extirpated north of Sacramento Valley by the 1950s (Gaines and Laymon 1984). Surveys throughout California in 1986–1987 found that only three areas in the state supported more than approximately five breeding pairs on a regular basis, including the Sacramento River between Colusa and Red Bluff, the South Fork of the Kern River, and the lower Colorado River (Johnson et al. 2008). In the 1999–2000 surveys, the Sacramento and Kern rivers were the only remaining areas with more than 1,000 hectares (2,470 acres) each of prime suitable habitat (i.e., high canopy cover, extensive understory, and structural diversity) (Halterman et al. 2003).

Within the Plan Area, the majority of CNDDDB records are from the Colorado River (CDFW 2013). Once considered abundant throughout the lower Colorado River, a dramatic decline of the species was noted during surveys in the 1970s and 1980s. The lower Colorado River and its tributaries supported an estimated 180–240 pairs in 1976–77. This population declined by an estimated 80% to 90% by 1986. In 1998, no pairs could be identified west of the Colorado River in the parts of California that had been occupied in 1976–77. Along the lower Colorado River and its major tributaries, losses have been greatest at lower elevations below 900 meters (3,000 feet) (Johnson et al. 2008).

Threats and Environmental Stressors

Western yellow-billed cuckoo is sensitive to habitat fragmentation and degradation of riparian woodlands due to agricultural and residential development (Hughes 1999), and major declines among western populations reflect local extinctions and low colonization rates (Laymon and Halterman 1989). Groundwater pumping and the replacement of native riparian habitats by invasive non-native plants, especially tamarisk, have substantially reduced the area and quality of available breeding habitats for yellow-billed cuckoo (75 FR 69222–

69294). Even where habitat is not degraded, the species has been extirpated from breeding areas occupied by four or fewer pairs (Laymon and Halterman 1987), possibly due to the inherent instability of small populations (Laymon and Halterman 1989). The extensive surveys in 1999 and 2000 found that large breeding populations in California only remain on the Sacramento and Kern rivers where there is still substantial prime habitat (Halterman et al. 2003). Non-native invasive species such as tamarisk (*Tamarix* spp.) may preclude use by western yellow-billed cuckoos; previously occupied willow-cottonwood habitats that converted to monotypic stands of tamarisk generally were no longer inhabited (Laymon and Halterman 1987), although Laymon (1998) reports two nest sites in tamarisk at the Bill Williams River site in Arizona. However, even at these sites, the habitat within the cuckoos' territories was still primarily willow-cottonwood (Laymon, pers. comm. 2012). Of the 33 known occurrences in the CNDDB database for the Plan Area, three of the sites were reported to have tamarisk invasion (CDFW 2013).

Pesticides may affect behavior of western yellow-billed cuckoo by loss of balance or may cause death by direct contact (Hughes 1999). Pesticides may contaminate preferred prey items, particularly lepidopteran larvae. In addition, some prey species, such as frogs, occur in pesticide-laden runoff adjoining agricultural land (Laymon and Halterman 1987). The western yellow-billed cuckoo also has shown pesticide effects on reproduction due to eggshell thinning (Gaines and Laymon 1984; Laymon and Halterman 1987). Of the 33 known occurrences in the Plan Area, agriculture (and associated access roads) adjacent to occupied habitat was reported to be a threat to five of the sites (CDFW 2013).

Yellow-billed cuckoos are also known to collide with windows, resulting in injuries and fatalities (Klem 1989, 1990). Whether this a substantial threat in the Plan Area is unknown, but it seems unlikely given the limited amount of development in occupied areas.

Climate change may be a stressor on yellow-billed cuckoos. Anders and Post (2006) examined BBS data for the eastern yellow-billed cuckoo for the period of 1966 to 2002 in relation to the North American Oscillation and El Niño Southern Oscillation climate systems. (The western yellow-billed cuckoo was excluded from the analysis due to few data.) Anders and Post (2006) found that

populations were sensitive to warm temperatures, with population declines in the year following the preceding breeding season with warm temperatures. They postulate that the decline in productivity is related to reduced available prey because they found that breeding only occurred in 2003 and 2004 on sites with more abundant prey. Lepidopteran larvae outbreaks appear to be more common during cooler weather (Anders and Post 2006). Further, it is possible that warmer temperatures cause earlier peaks of lepidopteran larvae that could be asynchronous with breeding by yellow-billed cuckoos at a time when prey is needed most (Anders and Post 2006).

Conservation and Management Activities

A rangewide conservation and assessment strategy for the western yellow-billed cuckoo is currently in preparation by a group of federal, state, and nongovernmental agencies organized by the Sacramento office of the USFWS (75 FR 69222–29294). Work on the conservation strategy is expected to be initiated in 2011.

Known occurrences of western yellow-billed cuckoo in the Plan Area are on BLM land. BLM Manual 6840 establishes Special-Status Species policy for plant and animal species and the habitat on which they depend (BLM 2001). The objectives of the BLM policy are:

- a. To conserve listed species and the ecosystems on which they depend.
- b. To ensure that actions requiring authorization or approval by the BLM are consistent with the conservation needs of special-status species and do not contribute to the need to list any special-status species, either under provisions of the ESA or other provisions of this policy (BLM 2001).

The BLM has identified the western yellow-billed cuckoo as a sensitive species and requires surveys in suitable habitat areas prior to authorizing activities that could disturb the species or its habitat.

Although the western yellow-billed cuckoo is not federally listed, several habitat conservation plans that would provide regulatory coverage for species, were it to be listed, have been implemented, including the Clark County Nevada Habitat Conservation Plan; the Lower Colorado River Multi-Species Conservation Plan; and the

California Department of Corrections Electrified Fence Project (for 26 sites throughout California, including nine sites in the Plan Area). Each of these conservation plans provides for conservation/protection and management of habitats that benefit the western yellow-billed cuckoo.

Wetland permits under Section 1600 of the California Department of Fish and Wildlife Code and federal Clean Water Act 404 issued by CDFW and the U.S. Army Corps of Engineers, respectively, also typically require avoidance, minimization, and mitigation measures for impacts to riparian habitats that may be used by western yellow-billed cuckoo and which may benefit the species. Further, any impacts to the species resulting in “take” are regulated by Section 2081 of the California Endangered Species Act, and full mitigation of impacts is required.

Data Characterization

Statewide systematic surveys for the western yellow-billed cuckoo have not been conducted since 1999 and 2000 (Haltermann et al. 2003), and there are only three recent (since 1990) known occurrences in the CNDDDB for the Plan Area (CDFW 2013). The current status of the species along the lower Colorado River and other areas where it has historically occurred, such as the Amargosa and Mojave rivers, is unknown. However, 26 of the 33 historic and recent known occurrences of the species are on public lands and are not subject to intense development pressure. The main concern for these areas is current habitat quality given that the western yellow-billed cuckoo requires large, dense tracts of riparian habitat. Water development (e.g., in the Victorville area) and invasive species such as tamarisk may have caused habitat degradation at some of the known occurrence sites since the cuckoo has been seen in the areas.

Management and Monitoring Considerations

Western yellow-billed cuckoo usually occur in large, dense tracts of riparian habitat, as summarized previously under Habitat Requirements. Therefore, management and monitoring will need to focus on maintaining, restoring, and enhancing large tracts of suitable habitat for the species, including controlling invasive species, such as tamarisk (Laymon and Haltermann 1985; Laymon 1998; Sogge et al.

2008) and ensuring water sources to maintain large riparian areas. The native, deep-rooted species that compose suitable cuckoo habitat, generally associated with perennial watercourses, require floods for maintenance and are tolerant of submersion when young (66 FR 38611–38626; Hughes 1999). Fire is also a consideration along the Colorado River, especially where people camp and may leave unattended camp fires (Comrack, pers. comm. 2011). The species is also highly dependent on adequate food sources (primarily caterpillars) for successful breeding (Martin 1987; Hughes 1999; Anders and Post 2006), so potential impacts on the prey base by pesticides applied to agricultural areas near suitable habitat are also a management concern. Pesticides may also cause lethal and sublethal poisoning to adults and young, adversely affecting the health and reproductive fitness of individuals and the viability of populations (Hughes 1999).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for western yellow-billed cuckoo, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 174,654 acres of modeled suitable habitat for western yellow-billed cuckoo in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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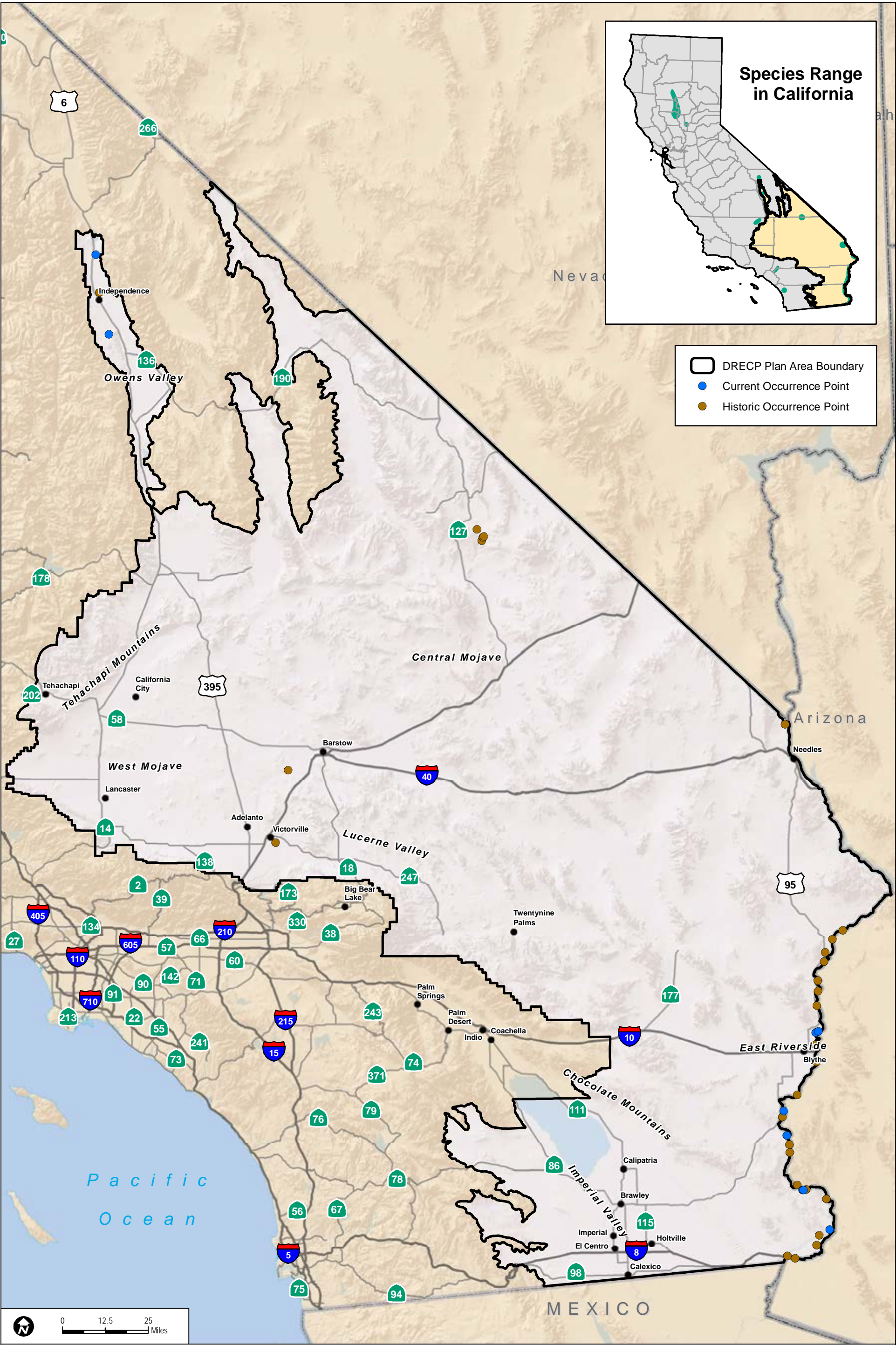
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BIRDS

Western Yellow-Billed Cuckoo (*Coccyzus americanus occidentalis*)

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B13

Western Yellow-billed Cuckoo Occurrences in the Plan Area

Desert Renewable Energy Conservation Plan (DRECP) Baseline Biology Report

October 2015

Willow Flycatcher (*Empidonax traillii*)

Legal Status

State: Endangered (willow flycatcher full species)

Federal: Endangered (southwestern willow flycatcher subspecies)

Critical Habitat: Designated on October 19, 2005 (70 FR 60886–61009) for southwestern willow flycatcher. The U.S. Fish and Wildlife Service (USFWS) proposed revised critical habitat on August 15, 2011 (76 FR 50542-50629), but the 2005 designation is still in place pending issuance of a final rule.

Recovery Planning: Final recovery plan (USFWS 2002) for southwestern willow flycatcher



Photo by Dudek.

Taxonomy

The willow flycatcher (*Empidonax traillii*) is a small passerine that was once considered along with the alder flycatcher (*E. alnorum*), as Traill's flycatcher (Grinnell and Miller 1944). Since 1973 the American Ornithological Union (AOU) has treated the alder flycatcher as a separate species and there are currently four recognized subspecies of *E. traillii*, three of which occur in California (*E. t. brewsteri*, *E. t. adastus*, and *E. t. extimus*) (USFWS 2002; Unitt 1987). Only the southwestern willow flycatcher subspecies (*E. t. extimus*) breeds in the Plan Area, and it is the primary focus of this account. The other two subspecies occur in the Plan Area only briefly during migration, and they are addressed in this account where relevant. The southwestern willow flycatcher was described by A. R. Phillips in 1948 from a collection by G. Monson from the lower San Pedro River in southwestern Arizona (60 FR 10695–10715). Southwestern willow flycatcher can be phenotypically distinguished from the other subspecies by its paler color, wing ratio, and song dialect (60 FR 10695–10715), although these are not reliable field identification

criteria (Sogge, pers. comm. 2012). Paxton (2000) concluded that the *E. t. extimus* subspecies is genetically distinct from the other subspecies, although intergrades between *E. t. adastus* and *E. t. extimus* have been reported (Unitt 1987).

Distribution

General

The willow flycatcher occurs throughout the United States with the exception of the extreme northeast and the southeast. In California, breeding populations of *E. t. adastus* and *E. t. brewsteri* are separated by the crest of the Sierra Nevada, while the historical range of *E. t. extimus* includes riparian habitats in the southern one-third of California, southern Nevada, Arizona, New Mexico, western Texas and northern Mexico (Sogge et al. 2010; USFWS 2002; Figure SP-B11), and, again, this is the only subspecies breeding in the Plan Area. The current range of *E. t. extimus* is similar to its historical range, the main difference being a reduction in the distribution and amount of existing suitable habitat within its historical range. This subspecies' breeding range extends as far north as the Santa Ynez River, Kern River, and the town of Independence on the Owens River (Craig and Williams 1998). Outside of California, historical breeding has occurred in southern Nevada, southern Utah, Arizona, New Mexico, and southwestern Colorado (Paxton 2000; Sogge et al. 2010).

Distribution and Occurrences within the Plan Area

Historical

Within the Plan Area, breeding southwestern willow flycatchers have been found at five general locations: Owens River Valley, Mojave River, San Felipe Creek (a tributary of the Salton Sea), the Lower Colorado River between Hoover and Parker, and the Lower Colorado River between Parker and the international boundary (Durst et al. 2008a). Willow flycatcher populations at these locations still exist, although numbers of territories have greatly declined at some locations, especially along the Colorado River (Durst et al. 2008a). These sites are discussed in further detail in the following section. There are no known general locations in the Plan Area that previously supported, but no longer support, southwestern willow flycatchers.

There are four historical (i.e., pre-1990) occurrences for southwestern willow flycatcher recorded in the Plan Area (CDFW 2013; Dudek 2013). The southwestern willow flycatcher occurrences are located north of Independence in Inyo County and in the vicinity of Mojave and California cities (Figure SP-B11).

Recent

As mentioned previously, there are five general locations in the Plan Area that currently support breeding populations of southwestern willow flycatchers. However, the southwestern willow flycatcher exhibits metapopulation dynamics with individuals commonly moving both among different sites within a breeding area and among different breeding areas (Sogge et al. 2010). Such movements reflect the dynamic interaction of suitable habitat and selection of breeding sites. In particular, small breeding sites are subject to variable use (Sogge, pers. comm, 2012). A detailed discussion of each of the five general breeding locations follows.

Owens River Valley: Most recently (as of 2007), Durst et al. (2008a) identified 28 territories at five sites in the Owens River Valley. However, almost all these territories occur north of the Plan Area. Within the Plan Area, two territories were located along the Owens River near Lone Pine in 1999, but the current breeding status at this location is unknown. Rourke et al. (2004) surveyed Hogback Creek near Lone Pine in 2001, but found no southwestern willow flycatchers. It is possible that none of the extant southwestern willow flycatcher territories found in the Owens River Valley occur within the Plan Area.

Mojave River: Durst et al. (2008a) stated that as of 2007, four nesting territories occur along the Mojave River near Victorville, but that territories are now gone from at least three other sites (Oro Grande, Upper Narrows, and Victorville Interstate 15). Nearby Holcomb Creek also once supported nest territories.

San Felipe Creek: San Felipe Creek is a tributary of the Salton Sea and as of 2007 supported four southwestern willow flycatcher nesting territories (Durst et al. 2008a).

Lower Colorado River – Hoover to Parker: As of 2007, Durst et al. (2008a) identified 14 territories remaining at six sites along this stretch

of the Colorado River. However, most of these territories occur at Topock Marsh on the Arizona side of the border. A California territory at Trampas Wash is considered extirpated (Durst et al. 2008a).

Lower Colorado River – Parker to South International Border: At one time, breeding southwestern willow flycatchers were located at 16 sites along this stretch of the Lower Colorado River, mostly on the Cibola and Imperial National Wildlife Refuges (NRWs). By 2007, the number of territories was reduced to one. McLeod and Koronkiewicz (2009) resurveyed this stretch in 2008 and “rediscovered” some territories (e.g., at Big Hole Slough), but territory numbers remain very low.

There are 101 recent (i.e., since 1990) occurrence records for willow flycatcher, of which the vast majority are identified only as willow flycatcher (CDFW 2013; Dudek 2013). There are five recent records for southwestern willow flycatcher along the Lower Colorado River in the stretch between the Cibola and Imperial NWRs, just south of where Interstate 10 crosses the river, and in the Havasu NWR area. There are also recent occurrences for southwestern willow flycatcher north of Niland east of the Salton Sea, in the Mojave River Narrows Regional Park, and in a tributary to the Owens River just above Tinemaha Reservoir. The remaining recent willow flycatcher occurrences are located in several regions of the Plan Area, including: Ridgecrest and the China Lake Naval Air Weapons Station, Amargosa Canyon, the Fremont Valley in the western Mojave, the southern Sierra Foothills west of Red Rock Canyon State Park, the Cities of Mojave and California City, Galileo Park north of 20 Mule Team Parkway, the southwestern portion of Edwards Air Force Base, the western portion of Mojave National Preserve, the Kingston Range, the Morongo Valley, Lake Tamarisk Golf Course in the Chuckwalla Valley, and north of Niland east of the Salton Sea.

Natural History

Habitat Requirements

In California, the southwestern willow flycatcher is restricted to riparian habitats occurring along streams or in meadows (Craig and Williams 1998; Sogge et al. 2010). As noted above under Distribution and Occurrences, there is a dynamic relationship between suitable

habitat and selection of breeding sites, with individuals commonly moving within general breeding areas and among different breeding areas. The structure of suitable breeding habitat typically consists of a dense mid-story and understory and can also include a dense canopy (60 FR 10695–10715). However, suitable vegetation is not uniformly dense and typically includes interspersed patches of open habitat. Typical plant species associated with their habitat include willow (*Salix* spp.), mulefat (*Baccharis salicifolia*), stinging nettle (*Urtica* spp.), cottonwood (*Populus* spp.), tamarisk (*Tamarix* spp.), and Russian olive (*Elaeagnus angustifolia*). Within the habitat structure parameters discussed above, southwestern willow flycatcher does demonstrate adaptability in that it can occupy riparian habitats composed of native broadleaf species, a mix of native and exotic species, or monotypic stands of exotics (Sogge et al. 2010). This subspecies is known to nest in monotypic stands of Russian olive and tamarisk (60 FR 10695–10715). Furthermore, along the San Luis Rey River in San Diego County, southwestern willow flycatcher has nested in riparian habitat dominated by coast live oak (*Quercus agrifolia*), and in Cliff-Gila Valley in New Mexico they are known to nest in tall box-elder. Plant species composition does not seem as important as a dense twig structure and an abundance of live, green foliage (Sogge et al. 2010). Also, the location of the nest seems to depend more on suitable twig structure and live vegetative cover than height or plant species composition (Sogge et al. 2010).

Riparian habitats within the Plan Area are also important stopovers to *E. t. adastus* and *E. t. brewsteri* as they migrate through (Finch and Kelley 1999). However, during migration willow flycatchers also use non-riparian habitats, including shrublands, grasslands, and agriculture (Finch et al. 2000). Other habitats used during migration typically lack the features associated with breeding sites, such as standing water, moist soils, and patch size and structure (Finch et al. 2000).

Southwestern willow flycatcher nesting sites are generally located near surface water or saturated soils (Table 1). Due to the variability of hydrologic conditions in Southern California, water availability at a site may range from inundated to dry from year to year or within the breeding season. Nonetheless, moisture levels must remain high enough to support appropriate riparian vegetation (Sogge et al. 2010). Dense

willow thickets are the most important habitat component for breeding *E. t. adastus* and *E. t. brewsteri* in California (Stefani et al. 2001).

Table 1. Habitat Associations for Southwestern Willow Flycatcher

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Dense Riparian	Breeding	Primary	Dense understory and mid-story	60 FR 10695–10715
Riparian	Foraging	Secondary	Openings within and edges of breeding habitat, over wet areas	Finch and Stoleson 2000

Foraging Requirements

Southwestern willow flycatchers are insectivorous and forage at the edges or internal openings of their territory, above the canopy or over open water. There are records of adults foraging outside of their territory and even within neighboring territories (Finch and Stoleson 2000). Their diet consists mainly of bees, wasps, flies, leaf hoppers, and beetles (Durst et al. 2008b), which they catch in the air, glean from vegetation, or occasionally pick, catch, or seize from the ground (Sedgwick 2000). However, because southwestern willow flycatcher is a generalist, its specific diet is difficult to describe. Diets can vary depending on the breeding site and weather conditions (Durst et al. 2008b). Presumably, the diet of migrating *E. t. adastus* and *E. t. brewsteri* is similar.

Reproduction

Southwestern willow flycatcher males and females become reproductively viable during their second year. This subspecies is predominantly monogamous although reports of polygyny are not uncommon (Sedgwick 2000). Males arrive at the breeding sites between early May and early June (USFWS 2002; Table 2). Females arrive 1 to 2 weeks after males and inhabit the territory of a male (Finch and Stoleson 2000). Nest building begins approximately 2 weeks after pair formation. Females build an open cup nest measuring 8 centimeters high by 8 centimeters wide (3.1 by 3.1 inches) with little to no assistance from the male.

The female incubates the eggs for an average of 12 to 13 days. The female provides the majority of care for the young; however, the male becomes more involved as the nestlings grow and demand more food. The nestlings fledge between 12 and 15 days after hatching (Sogge et al. 2010).

Southwestern willow flycatcher will typically renest following an unsuccessful attempt and less frequently may renest following a successful attempt. The clutch size of the first nesting attempt is typically three to four eggs but decreases with each new attempt (Ellis et al. 2008).

Studies in California along the South Fork Kern River showed that site fidelity for banded adults was 35.8% (Craig and Williams 1998); however, these studies did not differentiate between site fidelity and mortality. Studies in Arizona that only included surviving adults showed site fidelity as high as 66% as opposed to less than 50% for studies in the same area that did not take mortality into consideration (Luff et al. 2000). As *E. t. adastus* and *E. t. brewsteri* do not breed in the Plan Area, they are not addressed in this section.

Table 2. Key Seasonal Periods for Southwestern Willow Flycatcher

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Arrival					X	X						
Breeding					X	X						
Fledges						X	X					
Migration South								X	X			

Sources: 60 FR 10695–10715; USFWS 2002

Spatial Behavior

During their northbound and southbound migrations, other subspecies of willow flycatcher pass through areas occupied by nesting southwestern willow flycatchers. In Southern California, peak numbers of northbound *E. t. brewsteri* migrate the first couple weeks of June through occupied *E. t. extimus* breeding territories (Finch and Stoleson 2000). Therefore, for the purpose of focused surveys for

southwestern willow flycatcher, willow flycatchers occurring within the southwestern willow flycatcher breeding range can only be assumed to be southwestern willow flycatcher if detected between June 15 and July 20, when *E. t. brewsteri* have passed north to their breeding grounds (USFWS 2002). Willow flycatchers in the southwest migrate along riparian corridors (Finch and Stoleson 2000); because all three subspecies in California seasonally occur both north and south of the Plan Area, any riparian habitat within the Plan Area might represent important migration habitat for willow flycatchers. Finch and Kelley (1999) found that while migrating along the Rio Grande, willow flycatchers (including *E. t. extimus*) preferred habitats dominated by willows over other riparian species.

In adult southwestern willow flycatchers, movement to different breeding sites from year to year is not an uncommon occurrence and may occur as a response to low reproductive success at a particular nesting site. Distances covered range from 0.1 to 214 kilometers (0.06 to 133 miles) (Table 3). Year to year dispersal among juvenile birds is higher than in adults because juveniles rarely return to their natal site (Paxton 2007). Movement between breeding sites within the same breeding season typically occurs during pre- or post-breeding; although territory switching does occur, it makes up a small percentage of this type of movement (Paxton et al. 2007).

Territory sizes vary greatly depending on several factors, including but not limited to quality of habitat and population density. The observed range of territory sizes is about 0.1 to 2.3 hectares (0.3 to 5.7 acres), with most in the range of 0.2 to 0.5 hectares (0.5 to 1.2 acres) (USFWS 2002). Male territories tend to be larger before and after breeding. The area utilized within a territory tends to be smallest during incubation and when occupied by nestlings (Sogge et al. 2010).

Wintering locations for southwestern willow flycatcher are becoming better understood. Paxton et al. (2011a) combined information from mitochondrial DNA sequences and morphological characteristics from museum specimens collected for willow flycatchers from across their winter range and found that the Pacific lowlands of Costa Rica appear to be a key winter location for southwestern willow flycatcher, although Central American countries may also be important for the subspecies. Willow flycatchers will travel between 3,200 and 8,000 kilometers (2,000 and 5,000 miles) round-trip from their wintering

sites to their breeding sites. During migration, willow flycatchers use a greater variety of habitats, including some with non-riparian vegetation (Finch and Stoleson 2000).

Table 3. Movement Distances for Southwestern Willow Flycatcher

Type	Distance/Area	Location of Study	Citation
Breeding Territory	0.1–<2.3 hectares	California	USFWS 2002
Dispersal	0.1–214 kilometers	Arizona	Paxton 2007
Migration	3,200–8,000 kilometers	Throughout range	Finch and Stoleson 2000

Ecological Relationships

As is common for passerine bird species, southwestern willow flycatcher juveniles, eggs, and (less often) adults, are preyed upon by other birds, mammals, and reptiles. Predation is often the main factor responsible for nest failure (Sogge et al. 2010). In studies conducted along the lower Colorado River in 2003, depredation accounted for 57% of all documented nest failures (Koronkiewicz et al. 2004).

Brown-headed cowbirds (*Molothrus ater*), which are obligate brood parasites, parasitize the nests of several native passerine species, including southwestern willow flycatcher, and therefore also contribute to the overall nest failure for this subspecies. Female cowbirds lay their eggs in the nests of other bird species (host pair) at the expense of the reproductive success of the host pair (Finch and Stoleson 2000). Cowbirds have existed sympatrically with southwestern willow flycatcher throughout most of its range for hundreds of thousands of years. However, in Southern California, these two species have only co-occurred since 1900 (USFWS 2002). Nonetheless, the defense mechanisms used by southwestern willow flycatcher in Southern California in response to nest parasitism are similar to those used by willow flycatchers elsewhere, including nest abandonment (USFWS 2002) or burying the parasite egg in the nest floor (Finch and Stoleson 2000). Most southwestern willow flycatchers renest after abandoning their nest due to parasitism (USFWS 2002) and do not typically fledge flycatcher young from a parasitized nest (Sogge et al. 2010).

Despite evidence for parasitism, brown-headed cowbirds are not considered a primary threat to the success of the southwestern willow flycatcher (Sogge et al. 2010). This subspecies may be able to coexist with cowbirds as a stable population in the absence of other threats (USFWS 2002). Brown-headed cowbirds appear to be more of a threat at small, isolated nesting sites (Sogge et al. 2010). A study in coastal central California showed that individuals nesting in less-dense vegetation with a more open canopy are more likely to be parasitized (Finch and Stoleson 2000). Thus, high-quality, dense riparian habitat is valuable not only because it provides suitable habitat but also because it may reduce the ability for cowbirds to parasitize southwestern willow flycatcher nests.

There is no information on possible competition between migrating *E. t. adastus* and *E. t. brewsteri* and nesting *extimus* in the Plan Area, although it is possible that the groups compete briefly for the same food resources.

Population Status and Trends

Global: Declining (NatureServe 2011)

State: Critically Imperiled (NatureServe 2011)

Within Plan Area: Likely Declining

From the mid-1900s to the 1980s, populations of southwestern willow flycatcher declined rapidly (Unitt 1987). As of 2007, there were 1,299 known territories occurring within 288 breeding sites throughout the southwestern willow flycatcher's range. Of the 1,299 territories, 930 were surveyed in 2007 and the remaining 369 had been surveyed in 2006 or earlier (Durst et al. 2008a). Short-term studies on southwestern willow flycatcher have shown either a decline in population or no trend (Finch and Stoleson 2000). Within the Plan Area, significant declines have occurred along the Lower Colorado River, and occupied sites have declined in the Mojave River (Durst et al. 2008a). Overall, this subspecies is considered to be in decline (NatureServe 2011).

The majority of known territories and breeding sites occur in Arizona, New Mexico, and California. As of 2007, 96 breeding sites supporting approximately 172 territories have been documented in California, accounting for about 33% of all documented breeding sites in the

subspecies' range and 13% of all documented nesting territories for that year (Durst et al. 2008a). Arizona and New Mexico currently account for the majority of the documented breeding sites (57%) and documented territories (75%) (Durst et al. 2008a). In California, the largest populations are along the South Fork Kern River, the Owens River, San Luis Rey River, and Santa Margarita River (USFWS 2002); a portion of the Owens River occurs within the Plan Area (but few, if any, actual territories now occur within the Plan Area).

The other two California subspecies of willow flycatcher, *E. t. adastus* and *E. t. brewsteri*, have also suffered severe declines and consequently are also listed as endangered by the State of California. Intense agricultural and flood control activities in the Central Valley virtually eliminated the riparian habitat used by *E. t. brewsteri* (Serena 1982), and both *E. t. adastus* and *E. t. brewsteri* meadow habitats in the Sierra Nevada have been impacted by grazing (Stefani et al. 2001).

Threats and Environmental Stressors

The primary threat to the southwestern willow flycatcher is loss, modification, and fragmentation of suitable riparian habitat (Sogge et al. 2010). In general, increased human populations and development have resulted in a decline of riparian habitat, a habitat type that is naturally rare, patchy, and dynamic in the Southwest due to the varying hydrologic conditions of the region. The specific primary causes for loss and modification of riparian habitats have been dams and reservoirs, water diversion and groundwater pumping, channelization, flood control, agriculture, recreation, and urbanization (Sogge et al. 2010).

Impacts on suitable riparian habitat and conversion of adjacent native upland habitat have also resulted in indirect effects that are detrimental to this subspecies. Brown-headed cowbirds, discussed in the Ecological Relationships section above, are typically associated with anthropogenic influences, such as agriculture (cattle grazing), recreation (camp grounds and golf courses), and urbanization (lawns) (USFWS 2002). Although cowbird parasitism is not considered to be a primary threat to southwestern willow flycatcher, combined with other threats and stressors such as habitat loss and degradation, cowbird parasitism could be a significant contributor to population decline (USFWS 2002).

In California, the invasion of tamarisk and giant reed (*Arundo donax*) in riparian habitats has also been facilitated by anthropogenic disturbances (USFWS 2002). Although southwestern willow flycatcher is known to nest in monotypic stands of tamarisk, tamarisk is highly flammable and thereby has been suggested to pose a threat to southwestern willow flycatcher habitat (USFWS 2002; Finch and Stoleson 2000). However, while some territories have been lost in the last 20 years due to tamarisk fires, tamarisk has also supported many nesting territories, which have produced many hundreds of fledged flycatchers, which maintain and augment the population (Sogge, pers. comm. 2012). Additionally, Paxton et al. (2011b) concluded that using biocontrols such as tamarisk beetle (*Diorhabda* spp.) to eradicate tamarisk may negatively affect birds that have restricted distributions and sensitivity to seasonal defoliation, such as southwestern willow flycatcher, both in the short term and long term. Potential long term adverse and beneficial effects will be related to the rate regeneration and/or restoration of cottonwood and willow riparian habitats relative to the rate of loss of tamarisk. Therefore, for southwestern willow flycatcher, its relationship to tamarisk is more complex than tamarisk simply increasing fire risk (Sogge, pers. comm. 2012).

Giant reed forms large monotypic stands that are unsuitable for the subspecies (USFWS 2002) and are also subject to large fires. The risk of fire has also increased along streams where the flow of water has been reduced, due to dams or flood control, allowing for the accumulation of fuel in the understory (USFWS 2002).

Grazing, cowbirds, and water removal (Owens Valley) projects continue to be a threat to Sierra Nevada populations of *E. t. brewsteri* and *E. t. adastus* within their breeding range. Within the Plan Area, the same threats mentioned above for *E. t. extimus* would affect *E. t. brewsteri* and *E. t. adastus* where they impact riparian migration corridors.

Conservation and Management Activities

Survey, monitoring, and research efforts increased significantly after the southwestern willow flycatcher was federally listed as endangered in 1995 (60 FR 10695–10715). Since then, statewide surveys have been initiated in Arizona, New Mexico, and Utah. Breeding and migration ecology, demography, and habitat research has been conducted in Arizona, New Mexico, and California (e.g., Crag

and Williams 1998; Durst et al. 2008a, 2008b; Ellis et al. 2008; Hinojosa-Huerta et al. 2004; Langridge and Sogge 1997; Luff et al. 2000; Paxton et al. 2007; Sogge et al. 2010; Sogge and Paxton 2000). Range-wide population genetics work also has been conducted since the mid-1990s (USFWS 2002). Throughout the Southwest, several private, local, state, and regional efforts have formed in order to protect riparian habitats, including Partners in Flight and the Sonoran Bird Conservation Plan (USFWS 2002).

The Plan Area overlaps with the western part of the Lower Colorado River Recovery Unit, and the Basin and Mojave Recovery Unit identified in the recovery plan for southwestern willow flycatcher (USFWS 2002). The recovery plan sets forth alternative recovery criteria for the subspecies for downlisting to threatened and additional criteria for delisting (USFWS 2002). One recovery criterion (Criterion A) for downlisting to threatened status is increasing the known total population to a minimum 1,950 territories that are geographically distributed to allow metapopulation function and which are maintained over a 5-year period. An alternative criterion (Criterion B) for downlisting the subspecies to threatened is to increase the population to a minimum of 1,500 territories that are geographically distributed among management units and recovery units, protect the habitat supporting willow flycatcher populations from threats and loss, and maintain the population for a minimum 3-year period. The criteria for delisting the southwestern willow flycatcher is achieving Criterion A, providing protection from threats and creating/securing enough habitat to ensure maintenance of the populations and habitats over time (USFWS 2002).

The recovery plan also describes actions to offset habitat impacts, mitigation efforts, and other conservation efforts undertaken to the point in time the recovery plan was published in 2002. These conservation efforts included the following:

- Annual cowbird trapping on Marine Corps Base, Camp Pendleton, beginning in 1983, and annual surveys and nest monitoring started in 1999.
- Cowbird trapping, habitat restoration, and other conservation efforts in the Prado Basin area of the Santa Ana River beginning in 1996.

- Cowbird trapping and flycatcher monitoring and research associated with the construction of Isabella Dam.
- Management activities to benefit the southwestern willow flycatcher associated with the Roosevelt Dam in Arizona, including habitat acquisition, fencing, restoration, cowbird trapping, research, and monitoring.
- Protection and management of the Audubon Kern River Preserve, California, and habitat in the Cliff-Gila Valley, New Mexico, by the Nature Conservancy.

Several habitat conservation plans that provide regulatory coverage for southwestern willow flycatcher have been implemented, including the Clark County, Nevada, Habitat Conservation Plan; the Lower Colorado River Multi-Species Conservation Plan; the Western Riverside County Multiple Species Habitat Conservation Plan; the City and County of San Diego Multiple Species Conservation Programs; the San Diego Association of Governments North County Multiple Habitat Conservation Program; the Southern Orange County Habitat Conservation Plan; and the Sonoran Desert Multi-Species Conservation Plan. Each of these conservation plans provides for conservation/protection and management of riparian habitats that benefit southwestern willow flycatcher.

In 2005, the USFWS designated approximately 48,896 hectares (120,824 acres) of critical habitat for the southwestern willow flycatcher, including along the Mojave River in the Plan Area (70 FR 60886–61009). A proposed rule for revised critical habitat for the southwestern willow flycatcher was published in August 2011 (76 FR 50542-50629). Rather than designating aerial extent (i.e., total hectares) of critical habitat, as was done in the 2005 designation, the 2011 proposed rule expresses the total proposed critical habitat in terms of total stream length; approximately 3,364 stream kilometers (2,090 stream miles). The 2011 proposed rule designates the Mojave Management Unit, which includes a 35.7-kilometer (22.2-mile) segment of the Mojave River (which is substantially expanded downstream compared to the 2005 designation), a 11.2-kilometer (6.9-mile) segment of the West Fork Mojave River, a 19.6-kilometer (12.2-mile) segment of Holcomb Creek (outside the Plan Area), and a 20.0-kilometer (12.5-mile) segment of Deep Creek (which includes the Mojave River Forks Reservoir in the Plan Area, but most of which

is outside the Plan Area). The proposed rule also designates the Amargosa Management Unit segments, which include a 12.3 kilometer (7.7 mile) segment of the Amargosa River and a 3.5-kilometer (2.2-mile) segment of Willow Creek (3.5 km, 2.2 mi) in Inyo and San Bernardino counties. Neither of these two segments is in the current 2005 critical habitat designation.

Although the current 2005 critical habitat designation (nor the 2011 proposed designation) does not require specific conservation measures, it requires that evaluations of potential impacts on critical habitat be made on projects with a federal nexus (e.g., a federal permit action or funding) and may result in protection measures to avoid adverse modification or destruction of critical habitats associated with the project.

In 2010, the U.S. Geological Survey, in cooperation with the Bureau of Reclamation and the USFWS, developed a standardized survey protocol to be used for focused surveys throughout the range of the southwestern willow flycatcher (Sogge et al. 2010). This protocol provides information necessary to conduct and interpret survey results successfully, including a summary of basic ecological and population status information. Having a standardizing survey protocol allows for consistent data collection, reporting, and streamlined interpretation.

Restoration of breeding habitat for *E. t. adastus* and *E. t. brewsteri* has been a prime focus under the amended Sierra Nevada Forest Plan, and restoration efforts in the Owens Valley and near Mono Lake have improved breeding opportunities after original riparian nesting habitat was lost due to diversion of water to Los Angeles. All of these efforts are outside the Plan Area.

Data Characterization

At this time, information on the distribution and occurrence of the southwestern willow flycatcher within the Plan Area is limited, with very few documented occurrences. A greater level of confidence regarding the distribution of populations and isolated territories is needed in order understand the species' local status so that it can be managed adequately. Furthermore, the loss and degradation of riparian habitat is one of the most critical threats to the southwestern

willow flycatcher. More information is needed regarding the distribution of suitable and potentially suitable habitat within the Plan Area and potential impacts that may be occurring in those areas, such as occupancy by invasive species and hydrologic alterations. As recovery efforts continue and the population size increases, an important question for recovery and management is the potential for geographic expansion of the subspecies' breeding range.

Further investigation on the wintering grounds for southwestern willow flycatcher is needed in order to ensure that this subspecies is being protected adequately. Additional studies on the boundaries of the winter range and the quality of habitat used by this subspecies need to be conducted. Once this information is available, studies regarding the factors that limit survival of southwestern willow flycatcher during the winter can be conducted. Additionally, studies regarding threats to wintering grounds can be identified, followed by identification of methods needed, if any, to protect wintering grounds. Similar studies need to be conducted for migratory corridors used by this subspecies (Finch and Stoleson 2000). The same is true for *E. t. adastus* and *E. t. brewsteri*, especially in regard to how they use the Plan Area during annual migration periods.

Management and Monitoring Considerations

The recovery plan for the southwestern willow flycatcher outlines nine types of recovery actions: (1) increase and improve currently suitable and potentially suitable habitat; (2) increase metapopulation stability; (3) improve demographic parameters; (4) minimize threats to wintering and migration habitat; (5) survey and monitor; (6) conduct research; (7) provide public education and outreach; (8) assure implementation of laws, policies, and agreements that benefit the flycatcher; and (9) track recovery progress (USFWS 2002). As noted above, the Plan Area overlaps with portions of the Lower Colorado River Recovery Unit (Western Part) and the Basin and Mojave Recovery Unit. In the portion of the Lower Colorado River Recovery Unit overlapping the Plan Area, southwestern willow flycatcher occurrences are known from several locations south of Hoover Dam to the U.S.–Mexico border. In 2007, southwestern willow flycatcher territories were reported from the Hoover–Parker management unit and the Parker–Southern International Border

management unit (Durst et al. 2008a). In the portion of the Basin and Mojave Recovery Unit overlapping the Plan Area, southwestern willow flycatcher occurrences are known from the Mojave River in the Victorville area. In 2007, southwestern willow flycatcher territories were reported from the Owens Management Unit, Amargosa Management Unit, Mojave Management Unit, and the Salton Management Unit (Durst et al. 2008a).

Given the apparent limited occurrence of the southwestern willow flycatcher in the Plan Area, management for the subspecies should focus on removing existing or potential threats to riparian habitats, including invasive species, hydrologic changes in groundwater and surface water, and runoff from agriculture and urban uses. As discussed in Threats and Environmental Stressors, even though tamarisk is an invasive species, and ideally it would be eradicated and replaced with native cottonwood and willow habitats, tamarisk currently provides important nesting habitat for southwestern willow flycatcher (e.g., Paxton et al. 2011b; Shafroth et al. 2010). A temporal loss of tamarisk without available compensatory regeneration or restoration of native riparian habitat could have a substantial adverse effect on breeding southwestern willow flycatchers (e.g., Paxton et al. 2011b). Ellis et al. (2008), for example, recommends that tamarisk-dominant habitat in Arizona occupied by southwestern willow flycatcher should not be considered.

Ongoing monitoring and surveying efforts should continue in the Plan Area along the lower Colorado River, Mojave River, and Amargosa River and Willow Creek in areas containing suitable habitat in association with range-wide monitoring.

In addition to short-term cowbird control practices, such as trapping, long-term management practices may be needed for control of cowbird populations in southwestern willow flycatcher habitat if monitoring demonstrates that cowbirds are having significant local effects on southwestern willow flycatchers. Long-term management should emphasize reducing conditions known to attract cowbirds to riparian habitats, such as anthropogenic influences including golf courses, horse stables, and agricultural fields (Finch and Stoleson 2000; USFWS 2002). Providing educational programs for people residing near breeding populations would be beneficial in order to

reduce anthropogenic conditions that attract cowbirds and domestic pets that can prey on birds.

Because southwestern willow flycatcher habitat also is threatened by catastrophic wildfires, especially in areas that support tamarisk (Finch and Stoleson 2000), specific fire management plans should be prepared in coordination with local firefighters for discrete occupied habitat areas.

All of the above management considerations relative to riparian habitats would also benefit *E. t. adastus* and *E. t. brewsteri* where they migrate through the Plan Area.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for willow flycatcher, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 329,611 acres of modeled suitable habitat for willow flycatcher in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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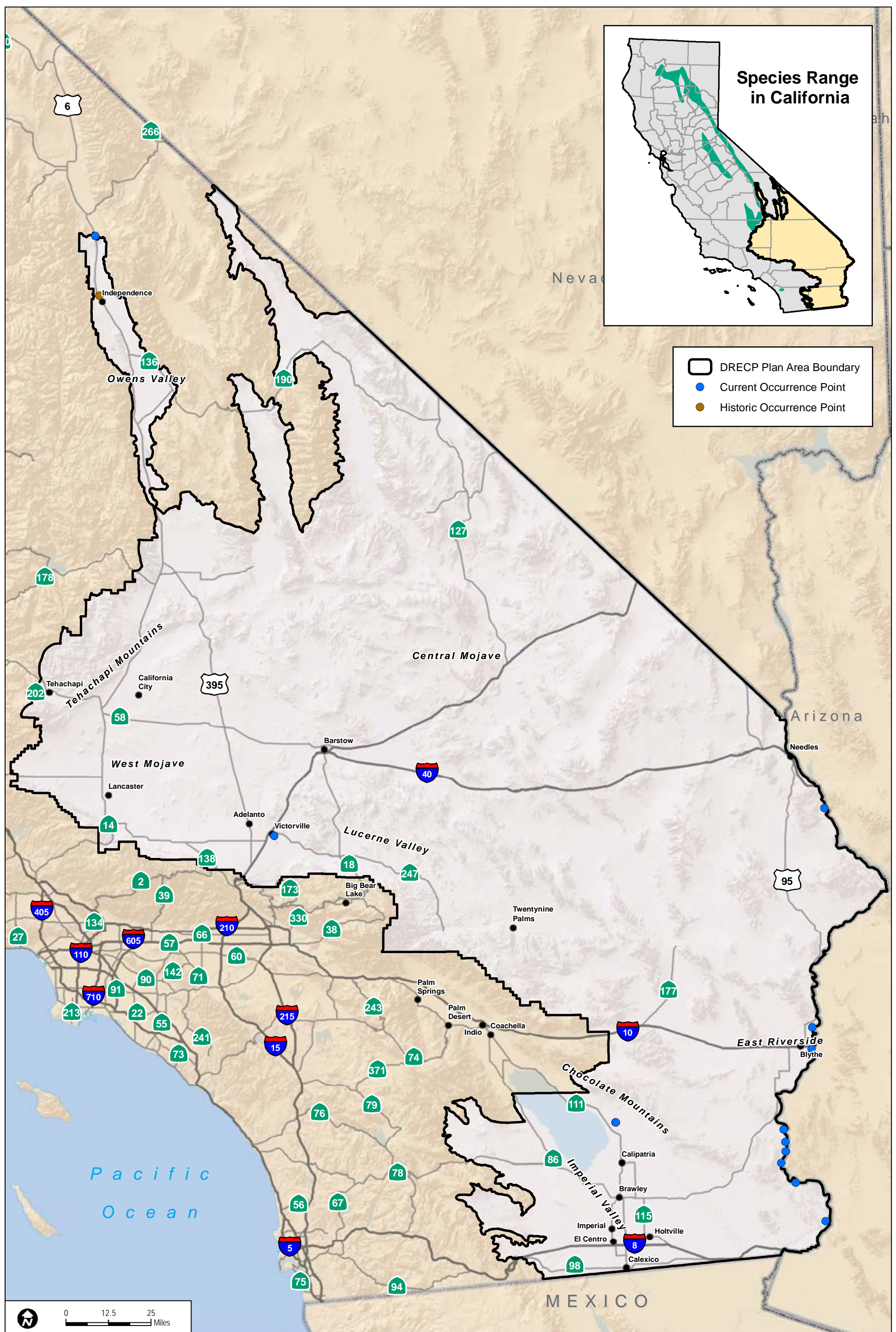
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Yuma Ridgway's Rail (*Rallus obsoletus yumanensis*)

Legal Status

State: Threatened,
Fully Protected

Federal: Endangered

Critical Habitat: N/A

Recovery Planning: A federal recovery plan for Yuma Ridgway's rail was completed on February 4, 1983, by the U.S. Fish and Wildlife Service (USFWS 1983). A Draft Revised Recovery Plan was published in February 2010 (USFWS 2010).



Taxonomy

In 1902, Herbert Brown described a clapper rail he had captured near Yuma, Arizona, as a light-footed clapper rail (*Rallus levipes*). In 1923, Dickey described it as a new species, the Yuma clapper rail (*Rallus yumanensis*) (Todd 1986; USFWS 2010), based on several minor morphological differences from other clapper rails, as well as its isolated range and freshwater habitats (Banks and Tomlinson 1974).

Although there was some subsequent controversy over the rail's classification (Van Rossem 1929; Oberholser 1937), for more than 60 years it had been widely treated as a subspecies of *R. longirostris* (i.e., *R. longirostris yumanensis*) (Fleischer et al. 1995). However, recent genetic studies evaluating the relatedness of rails split clapper rails into three species. Currently, Ridgway's rail (*Rallus obsoletus*) includes the "California" (*R. o. obsoletus*), "Yuma" (*R. o. yumanensis*), and "light-footed" (*R. o. levipes*) subspecies, and others in Mexico. Any bird formerly known as "clapper rail" observed in California, Nevada, or Arizona is now this species. The name "clapper rail" was retained for the birds on the east coast of the United States, but its scientific name has changed (ABA 2014).

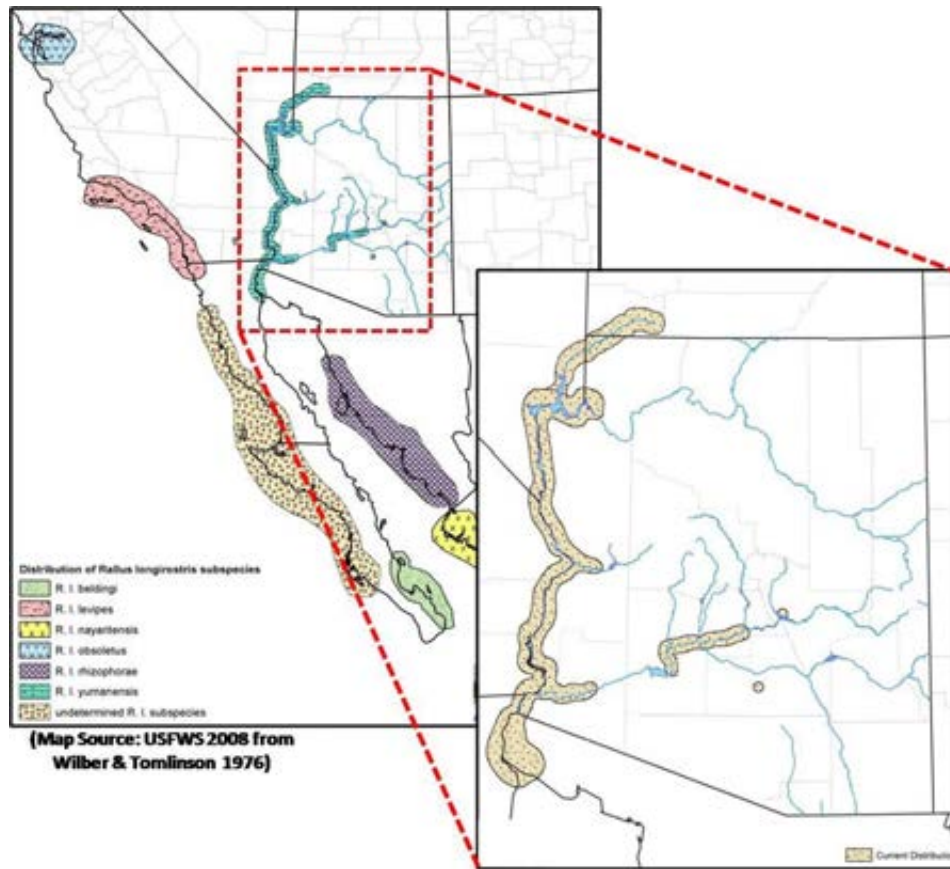
A description of the species' physical characteristics can be found in the Draft Revised Recovery Plan (USFWS 2010).

Distribution

General

Yuma Ridgway's rail breeds along the lower Colorado River (including La Ciénega de Santa Clara in Mexico), the Gila River drainage in Arizona, Lake Mead (and the Overton Arm) and its local tributaries, the Virgin River in Nevada and Utah, and the Salton Sea/Imperial Valley areas of California. Figure 1 shows the general breeding range of the species and Figure SP-B14 indicates known occurrence in the Plan Area. In the Plan Area, the main habitat areas for this subspecies are located along the Colorado River and around the Salton Sea (including Dos Palmas Springs).

There are at least three "outlier" observations for Yuma Ridgway's rail. In 1977, an individual was identified by vocalization on several days at Harper Lake northwest of Barstow (Figure SP-B14) but was not observed subsequently and was considered to be an unpaired individual (CDFW 2013). In 1978, Yuma Ridgway's rail was identified at Cronese Lake in the central Mojave (Garrett and Dunn 1981). In 1989, a single Yuma Ridgway's rail was observed at the Ash Meadows National Wildlife Area located about 90 miles northwest of Las Vegas.

Figure 1. Range of Yuma Ridgway's Rail

Distribution and Occurrences within the Plan Area

Historical

The historical distribution of Yuma Ridgway's rail is unclear. Todd (1986), in an extensive investigation of Yuma Ridgway's rail literature, reported that rails were first observed by J.G. Cooper near Fort Mojave in 1884. This is likely the earliest record. However, Joseph Grinnell performed an extensive survey of the Colorado River between Needles and Yuma in 1914 and did not record any observations of this species. However, he later documented the Yuma Ridgway's rail from the lower Colorado River (Grinnell and Miller 1944, cited in Todd 1986). The Desert Renewable Energy Conservation Plan (DRECP) Area includes eight historical (i.e., pre-1990) records of the Yuma Ridgway's rail in the California Natural Diversity Database (CNDDB) and others located just outside the Plan Area (Figure SP-B14) (CDFW 2013). Several of the historical

occurrences occur along the lower Colorado River south of Parker to about 22 miles north of Yuma, Arizona (Figure SP-B14). Historical occurrences are also located at the Salton Sea, along the All American Canal, the New River, and the Holtville main drain in the Imperial Valley, as well as the single record each at Harper Lake in 1977, and Cronese Lake in 1978 (Figure SP-B14).

Yuma Ridgway's rail appears to respond positively to human activities that create habitat. Construction of dams both on the Colorado River and along adjacent tributaries has possibly contributed to the shift in the Yuma Ridgway's rail's distribution (Anderson and Ohmart 1985; Ohmart and Smith 1973). Table 1 shows the relationship of upstream distribution of the Yuma Ridgway's rail in relation to water management activities. These dams have the effect of creating sedimentation and backwater areas, thus providing additional shallow-water emergent habitat required by the Yuma Ridgway's rail (CVCC 2007). Near the edge of the Salton Sea freshwater marsh ponds have been built and maintained to create habitat that now supports Yuma Ridgway's rails.

Table 1. Upstream Distribution of Yuma Ridgway's rail and Relationship to Dam Construction and the Salton Sea Flood Event

Location	Year completed	Year Yuma Ridgway's rail first found
Salton Sea	1905 (flooded)	1931
Laguna Dam	1905	1921
Headgate Dam	1941	1946
Parker Dam	1938	1954
Topock and Upper Lake Havasu	1938	1966
Needles Area	—	1982
Hoover Dam	1936	1986
Virgin River	—	1998
Source: USFWS 2010		

Recent

The recent (i.e., since 1990) documented distribution of Yuma Ridgway's rail in the Plan Area is similar to the historic distribution,

but with some apparent shift along the Colorado River. The distribution now ranges from about Lake Havasu to near Yuma, Arizona (Figure SP-B14). The recent distribution in the Salton Sea/Imperial Valley area is similar to the historic distribution. The Coachella Valley Habitat Conservation Plan (CVCC 2007) reports that Yuma Ridgway's rail is found on Salt Creek and the Dos Palmas oasis in the southern Coachella Valley. The CNDDB contains 37 records for the period between 1990 and 2010 (CDFW 2013) and the USFWS database includes 20 records from 2004 to 2010 (USFWS 2011). The records from the USFWS database are located around the eastern edge of the Salton Sea, south of El Centro, and along the Colorado River near the Colorado River Indian Reservation and near the Imperial Reservoir. (It appears that there is some overlap between the USFWS and CNDDB databases for the period from 2004 to 2010, but the USFWS database contains the most recent data from USFWS protocol surveys.)

Yuma Ridgway's rail has also colonized Ash Meadows National Wildlife Refuge (NWR) and has established a resident population there. Yuma Ridgway's rail has also been known to inhabit Wixom Marsh near Seeley in the Imperial Valley. A May 2007 survey detected Yuma Ridgway's rails defending breeding territories, and a Yuma Ridgway's rail was heard calling in the marsh in January 2013. The marsh is thought to support two breeding territories.

Natural History

Habitat Requirements

Among the subspecies of clapper rail, only *yumanensis* is known to breed in freshwater marshes. By far, the preferred habitat consists of cattails (*Typha* spp.) and bulrush (*Scirpus* spp.) (Anderson and Ohmart 1985; Todd 1986; Eddleman 1989). Eddleman (1989) found that habitat use by the subspecies on two study sites varied somewhat over different seasonal periods (i.e., early breeding, late breeding, post-breeding, early winter, and late winter), but that some combination of cattail and bulrush accounted for the majority of the observations across all periods. Combining data from the two study sites, use of cattail/bulrush habitats ranged from 66% of observations in the post-breeding period to 86% in the early breeding period

(Eddleman 1989). Notably, on one of the sites, rails were observed in tamarisk (*Tamarix* spp.) second-most frequently behind cattail, with a range of 11% of the observations in the late winter period to 37% in the post-breeding and 36% in the early winter periods (Eddleman 1989). USFWS (2010) notes that the subspecies has been observed in shoreline areas with a mix of trees, including willow (*Salix* spp.) and tamarisk. However, although they are occasionally observed under the woody vegetation fringing a freshwater marsh, woody vegetation doesn't hold much habitat value for Yuma Ridgway's rail compared to marsh vegetation (i.e., cattails and bulrushes).

Optimum habitat for Yuma Ridgway's rail results from a complex interplay of water levels, appropriate vegetation and vegetation characteristics (e.g., matting, dry areas, senescence), the timing of seasonal flooding, and possibly the timing of crayfish (*Procambarus clarkii* and *Orconectes virilis*, its primary prey) reproduction (Bennett and Ohmart 1978; Todd 1986). In a draft Recovery Plan for Yuma Ridgway's rail, the USFWS (2010) characterized optimum habitat as consisting of:

“... a mosaic of emergent vegetation averaging greater than 2 meters (6 feet) high (Anderson and Ohmart 1985; Eddleman 1989), shallow (less than 30 centimeters [12 inches]) open water areas either as channels or pools with minimal daily water fluctuation (Tomlinson and Todd 1973; Gould 1975), open dry ground (slightly higher than the water level) between water, vegetation, or marsh edge for foraging and movement (Gould 1975; Anderson and Ohmart 1985; Eddleman 1989; Conway et al. 1993), and a band of riparian vegetation on the higher ground along the fringes of the marsh that provides cover and buffer areas that may be used seasonally (Eddleman 1989).”

An overriding consideration for nesting by Yuma Ridgway's rail is that the nest substrate be stable (Eddleman 1989; USFWS 2006, 2010). Sparsely vegetated areas are more likely to be occupied if crayfish are abundant (Anderson and Ohmart, 1985). Yuma Ridgway's rail depends on a continuous source of water, most likely because crayfish are similarly dependent. However, the species also seems tolerant of seasonal fluctuations in water level that characterize the Colorado

River (Eddleman 1989), as long as the change in level is not too abrupt (Conway and Eddleman 2000, cited in USFWS 2010). Similarly, Gould (1975) suggested that short-term changes in water level should be avoided. Rails may have several nests and can move eggs to nests that are less threatened if need be, but if the habitat dries out, rails will abandon the area (Bennett and Ohmart 1978; Johnson and Dinsmore 1985).

According to Gould (1975), in addition to the basic habitat requirements of standing water and marshland vegetation, the following habitat parameters are desirable to support high Yuma Ridgway's rail densities:

1. "Water flowing through many small channels, from 0.5 to 3 meters (1.5 to 10 feet) wide either covered by vegetation or appearing as open water or appearing as small bodies of open water, 0.02 to 0.2 hectare (0.05 to 0.5 acre) in size.
2. Extensive areas of water where depth is less 0.3 meter (1 foot). Little or no daily fluctuation in water level.
3. High ground found in strips, or less importantly as small isolated islands.
4. Emergent vegetation being cattail and bulrush with little or no carrizo cane [aka, giant reed (*Arundo donax*)]. In areas of carrizo cane, stem density is generally too high and there are few down stems."

An important aspect of Yuma Ridgway's rail habitat is that over time, without occasional scouring by seasonal floods, marshes tend to become both overgrown (e.g., stem density too high), and much of the open or semi-open water fills with mats of old vegetation. The effects of this maturing process, or senescence, are that it becomes impossible for rails to move through vegetated habitat areas compared to open or semi-open aquatic habitat. Thus, foraging efficiency decreases as the habitat becomes choked with vegetation matting (Hinojosa-Huerta et al. 2008).

Foraging Requirements

Principal prey of Yuma Ridgway's rail are the two introduced species of crayfish that occur in the area (Inman et al. 1998). Ohmart and

Tomlinson (1977) found that about 95% of the stomach contents of two Yuma Ridgway's rail specimens were crayfish, leading them to suggest that the range shift of Yuma Ridgway's rail may have been facilitated by the introduction and spread of the crayfish. Other prey items taken by Yuma Ridgway's rail include small fish, insects, amphibian larvae, clams, and other aquatic invertebrates (Todd 1986; USFWS 2010).

Reproduction

Yuma Ridgway's rail begins breeding activities in the early spring, usually in March or early April (Eddleman 1989), although mating calls may be heard as early as February (USFWS 2010). Breeding begins with the establishment of breeding territories. Birds occupying more peripheral territories may mate a month or so later (Arizona Game and Fish Department 2007). Both males and females vigorously defend territories. Nesting occurs from March through May, but can vary with location and annual seasonal rainfall patterns (USFWS 2010).

Observed clutch sizes for 15 Yuma Ridgway's rail's nests in the lower Colorado River and Salton Sea ranged from 5 to 8 eggs (Eddleman and Conway 2012). Incubation was observed to last 23 to 28 days at nests in Arizona (Eddleman and Conway 2012). Both males and females incubate the eggs, with males incubating during the night shift and females incubating during the day (Eddleman 1989). Hatching success is high but juvenile mortality is also high (Bennett and Ohmart 1978; Eddleman 1989).

Young are precocial and within about 2 days of hatching they accompany adults on foraging trips, learning quickly to capture their own prey (Hunter et al. 1991). Family groups stay together for about 1 month, after which time the chicks separate from the parents. First flight occurs about 60 days after hatching (Arizona Game and Fish Department 2007).

Although nests may be from 6 centimeters (approximately 2.5 inches) to over 1 meter (approximately 3.3 feet) above the water level (average = 19.8 centimeters [approximately 7.8 inches]) (Eddleman 1989), as water levels rise, the birds may raise the level of existing nests or move eggs to a different nest. Consequently, Yuma Ridgway's

rail may have several nests available for use (Conway and Eddleman 2000, cited in USFWS 2010).

Spatial Behavior

Migration and dispersal patterns of Yuma Ridgway's rails are not well understood. The current scientific thinking is that Yuma Ridgway's rails do not migrate seasonally. However, post breeding dispersal is likely possible over long distances. It was first assumed that Yuma Ridgway's rail migrated south during the winter (Smith 1974; Todd 1986), but Eddleman (1989) observed that up to 70% of the populations he studied remained at their site year-round in the lower Colorado River area. Also, as noted in Distribution and Occurrences, the observations for Yuma Ridgway's rail at Harper Lake northwest of Barstow in 1977 (CDFW 2013), another at Cronese Lake in 1978, an unpaired individual at Ash Meadows National Wildlife Area in 1989 (Garnett et al. 2004), and the finding in 2013 of an individual at a desert solar project located 32 miles from the nearest occupied habitat indicate that Yuma Ridgway's rails are capable of long-distance movements. The purposes, frequency, and distances involved in long-range movements by Yuma Ridgway's rails remain unclear, and is an important topic for future research (USFWS 2006, 2010).

Yuma Ridgway's rail also shows seasonal variability in its use of habitat and in its home range size (USFWS 2010). According to Eddleman (1989), there are five movement patterns by Yuma Ridgway's rail outside of their breeding territory:

- Dispersal by juveniles
- Dispersal during the breeding season by unpaired males
- Movements of post-breeding adults
- Movements during late winter
- Home-range shifts associated with high water

The triggers for these movements appear to be the need to find suitable habitat (juvenile dispersal, post-breeding movements, late winter movements), the need to find mates (late winter movements, movements of unpaired males during the breeding season), and/or the need to locate food (post-breeding and late winter movements)

(Eddleman 1989). Home ranges are variable over different seasons, ranging on average from 7 to 8 hectares (17 to 20 acres) in the early and late breeding periods, to 15 hectares (37 acres) in the post-breeding period, and 24 hectares (59 acres) in the late winter period (Conway et al. 1993). Females have larger ranges than males in the post-breeding period at 21 hectares (51 acres), compared to 9 hectares (22 acres), but the two sexes have similar home range sizes the rest of the year (Eddleman 1989).

Ecological Relationships

Yuma Ridgway's rail is prey for several species, including coyote (*Canis latrans*), common raccoon (*Procyon lotor*), great horned owl (*Bubo virginianus*), Harris' hawk (*Parabuteo unicinctus*), and northern harrier (*Circus cyaneus*) (USFWS 2010). Eddleman (1989) attributed 36 out of 37 known mortalities from natural causes to predation (50% by mammalian predators, 22% by avian predators, and 28% by unknown predators). Because these predators are generalists, however, the rail probably is not a critical element of their diets and likely is taken opportunistically.

As discussed previously, suitable habitat for Yuma Ridgway's rail depends on water levels, appropriate vegetation, the timing of seasonal flooding, and possibly the timing of crayfish reproduction. The subspecies appears to be particularly sensitive to water levels and may have several nests and can move eggs to nests that are less threatened by rising water levels if need be.

Population Status and Trends

Global: Vulnerable (NatureServe 2010)

State: Critically imperiled (NatureServe 2010)

Within Plan Area: Critically imperiled (NatureServe 2010)

Yuma Ridgway's rail in the United States has shown recent range extensions northward from the Colorado River Delta and the southern end of the Colorado River into Lake Mead and the Virgin River, indicating that the species is reproducing enough to support such a range shift (USFWS 2006, 2010). The species' first recovery plan (USFWS 1983) indicated that the breeding population had been stable for 10 years at the desired level of 700 to 1,000 individuals. As a

result, a down-listing package was prepared for the Federal Register in 1983. However, subsequent flooding of important habitat on the lower Colorado River resulted in the proposal not being published (USFWS 2006).

The long-term assessment of population trends is complicated by several factors identified by the USFWS (2010), including:

- Inconsistencies in the proportion of suitable habitat surveyed in different years; and
- Different survey protocols, such as playback methods (e.g., continuous vs. intermittent call playback), seasons of surveys, and differing levels of surveyor experience.

While the data for the United States populations of Yuma Ridgway's rail do not allow for statistical population estimates, they do provide minimum number of rails in the census areas, which is the actual count of rails detected on survey routes, and which represents some subset of the actual population. Between 2000 and 2008, the minimum numbers in the United States ranged from 503 individuals in 2000 to 890 individuals in 2005 (USFWS 2010, Table 1). In the Plan Area, including the Colorado River and Salton Sea, the range over this same period was 472 individuals in 2001 to 849 individuals in 2005. The 2008 minimum number was 592 individuals along the Colorado River and at the Salton Sea (USFWS 2010). Within the lower Colorado River Delta region of Mexico (Ciénega de Santa Clara), Hinojosa-Huerta et al. (2008) documented a decline of 55% for the period of 1999 to 2002, but there was no statistically significant change between 1999 and 2006. The population was estimated to be 5,974 individuals (95% Confidence Interval = 4,698–7,482) in 2006, making it the largest documented population of the Yuma Ridgway's rail.

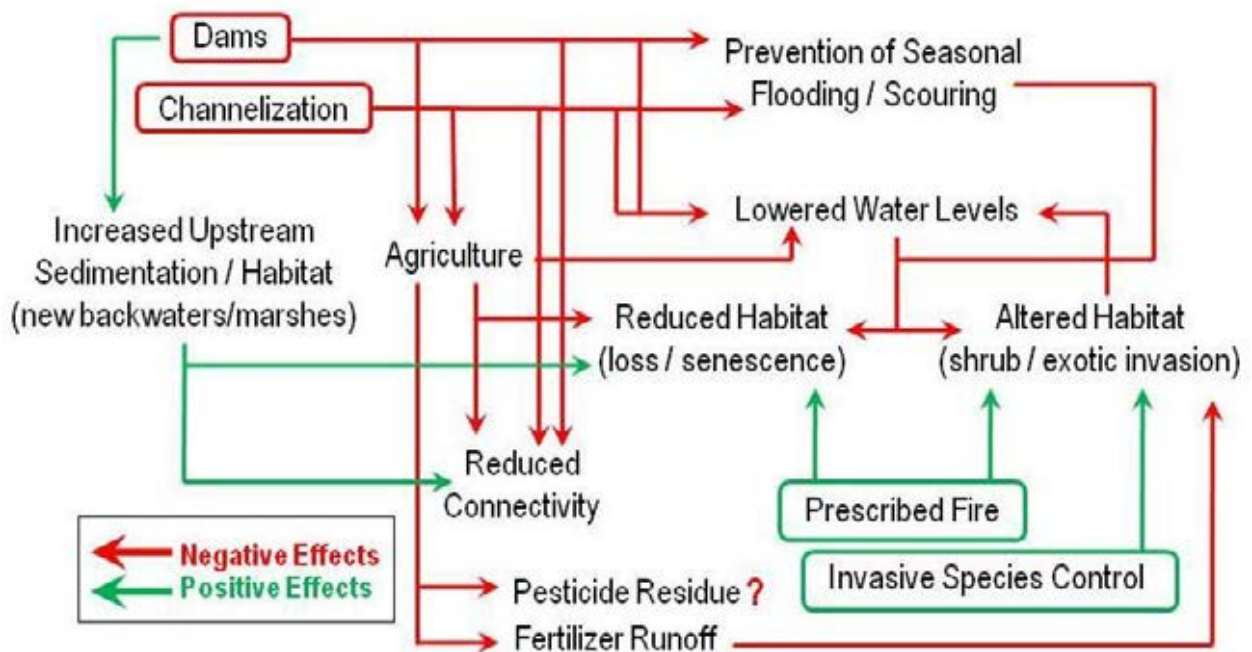
Threats and Environmental Stressors

Habitat destruction and modification is the primary threat to Yuma Ridgway's rail (USFWS 2010). The natural hydrologic regime along the lower Colorado River has been altered by damming, channelization, and bank stabilization, the last of which has separated the main river channel from backwater and floodplain areas where marsh habitats would naturally form (USFWS 2010). While damming has likely created additional marsh habitat for rail in some areas, the

dams have resulted in altered flood regimes from historical seasonal winter and spring flooding events that are necessary to maintain healthy marsh systems. These natural flooding events would have removed much of the thick matting of dead vegetation and build-up of sediments that allow for efficient foraging and escape from predation. Without active management, the value of these marsh habitats for Yuma Ridgway's rail is reduced, and the habitat may disappear altogether (USFWS 2010). On the other hand, dams have also resulted in sedimentation of ancillary streams and creeks upstream, thereby increasing the extent of backwaters and marshes available for the Yuma Ridgway's rail. This creation of new habitat has been cited as one reason for the shift of the species' range upstream (see Distribution and Occurrences within the Plan Area).

Figure 2 presents a generalized conceptual model of water management (dams, channelization) and their potential negative and beneficial effects on marsh habitat for the Yuma Ridgway's rail.

Figure 2. A Generalized Conceptual Model for the Effects of Water Management on Yuma Ridgway's Rail



Currently, the marshes at the Salton Sea Sonny Bono National Wildlife Refuge (NWR) and Imperial Wildlife Area are managed by flushing salts from the wetland ponds. Use of water for management of clapper rail habitat has increased since 2004 and may be constrained in the future by competing water uses, such as agriculture, that may increase the cost and availability of water (USFWS 2010).

Environmental contaminants may also pose threats to the species. Eddleman (1989) documented high levels of selenium in the Yuma Ridgway's rail, its eggs, and its primary food source (i.e., crayfish). Similar levels of selenium were responsible for reproductive damage in mallards (*Anas platyrhynchos*) (Lemly and Smith 1987, cited in Eddleman 1989). Several studies have found high concentrations of selenium in the Colorado River and the Salton Sea (Andrews et al. 1997; King et al. 2000; Rusk 1991, cited in USFWS 2010). In the discussion of these studies, USFWS (2010) stated, "selenium levels in those studies were high enough to indicate the potential for exposure and adverse effects to Yuma Ridgway's rails." Also, "... based on the available data, we do identify it [selenium] as a long-term threat to survival and recovery" (USFWS 2010, p. 16).

Conservation and Management Activities

A Yuma Ridgway's rail Recovery Team was created in 1972 (USFWS 2006, 2010) that instituted survey protocols and additional research on the species. A formal recovery plan was created in 1983 and some of the recommended recovery actions commenced. Following these initial studies, the recovery team became inactive except for the coordination of annual surveys completed by volunteers from state and federal agencies (USFWS 2010).

In 1995, a group composed of local, state, and federal agencies; water and power agencies; environmental and recreational groups; and Native American tribes was formed to develop the Lower Colorado River Multi-Species Conservation Program (LCRMSCP). In December 2004, the LCRMSCP was completed (LCRMSCP 2004). Covering 26 species, including the Yuma Ridgway's rail, the LCRMSCP calls for the creation of an additional 512 acres of Yuma Ridgway's rail habitat and its management in an adaptive management framework to not only protect Yuma Ridgway's rail but also to understand how the management of threats and stressors affects Yuma Ridgway's rail abundance.

Other programs to protect and enhance Yuma Ridgway's rail habitat have been created at the Salton Sea by the U.S. Bureau of Reclamation (USFWS 2002) and at the Torres Martinez Desert Cahuilla Indians in 2005. Prescribed fire has been used to enhance Yuma Ridgway's rail habitat at the Sonny Bono, Havasu, and Imperial NWRs, as well as the Mittry Lake Wildlife Area (USFWS 2010).

In 2006, a 5-year review of the recovery plan was completed (USFWS 2006), and the following five actions were recommended:

- Revise the recovery plan.
- Involve USFWS with the protection of the Ciénega de Santa Clara (Mexico), ensuring a continuous water source for this highly significant sub-population, which, based on the 2006 population estimate by Hinojosa-Huerta et al. (2008) accounts for approximately 87% of the known Yuma Ridgway's rail population.
- Establish new survey protocol and training using an adaptive management scenario to determine the effectiveness of management actions.
- Develop or revise management plans for the National Wildlife Refuges and State Wildlife Areas focusing on areas of declining Yuma Ridgway's rail populations and habitat quality.
- Continue to support research efforts into the Yuma Ridgway's rail, especially the possible effects of elevated selenium levels.

The federal government initiated efforts to implement these recommendations in 2007 (USFWS 2010). In February 2010, a Draft Revised Recovery Plan was released for public review (USFWS 2010). In this revision, the strategies used for the continued persistence of the Yuma Ridgway's rail focused on "... providing long-term management and protection for a sufficient amount of core and other habitats to support a viable population of Yuma Ridgway's rails, monitoring of populations and habitats, research to provide effective conservation and recovery, and application of research results and monitoring through adaptive management" (USFWS 2010, p. iv).

Data Characterization

Numerous surveys have been conducted for the Yuma Ridgway's rail throughout its range in the U.S. and the Plan Area. Table 1 of the Draft Revised Recovery Plan shows that surveys were conducted along the lower Colorado River and at the Salton Sea every year from 1969 to 2007 (USFWS 2010). These data are not appropriate for estimating population sizes for various reasons, as discussed in Population Status and Trends, but they do provide information for the actual number of individuals observed along survey routes and allow some insight into occurrence population fluctuations and trends from year to year.

Despite the annual surveys and a reasonably good understanding of suitable habitat characteristics, information gaps that would inform management still exist, and research into the following topics should be conducted:

- The effects of elevated levels of selenium and pesticide residue on Yuma Ridgway's rail reproduction and survival.
- The extent and importance of seasonal migration.
- Re-nesting.
- The effects of prescribed fire on senescent marshes and Yuma Ridgway's rail.
- The possible effects of increases in opportunistic predators associated with human presence and development (e.g., coyotes, feral pets, common raven [*Corvus corax*]).
- Genetic structure and gene flow.
- Seasonality and population structure of crayfish.
- The effects of human activities on Yuma Ridgway's rail, including noise, lighting, human presence, wildfire, and power lines.

Management and Monitoring Considerations

Management for Yuma Ridgway's rail should focus on maintaining high-quality marsh habitat. This includes not only the amount of available habitat, but the need for addressing water management issues important for maintaining high-habitat quality. Such issues include controlling water flows; establishing appropriate seasonal flooding

and/or prescribed fire regimes to prevent decline and overgrowth of marshes; controlling of water levels during nesting periods; maintaining a habitat mosaic, that includes some upland areas; controlling exotic invasive species (e.g., tamarisk, giant reed); and controlling potentially harmful chemicals and other pollutants.

All management actions should be in concert with the goals of the Draft Revised Recovery Plan (USFWS 2010), which includes the following specific recommendations:

1. Define the minimum population size that must be maintained for the Yuma Ridgway's rail in the U.S. to achieve recovery and document progress toward meeting that population size.
 - Determine the number of breeding birds in the U.S. that provides for a statistically and genetically secure population.
 - Conduct coordinated surveys for Yuma Ridgway's rail in the U.S. to document when minimum viable population levels are met.
2. Define the physical parameters of and document the amount of Yuma Ridgway's rail habitat in the U.S. needed to support the minimum viable population size.
 - Refine knowledge of rail use of habitats that support determination of the total amount of habitat needed in the U.S.
 - Develop techniques for managing habitats to maintain suitable conditions for Yuma Ridgway's rail.
 - Complete an assessment of the amount and location of Yuma Ridgway's rail habitat in the U.S. every 5 years.
3. Ensure that existing and new habitats for Yuma Ridgway's rail are protected and managed for long-term habitat suitability.
 - Develop and implement management plans for all important federal- and state-owned core areas to maintain suitable habitat conditions.
 - Ensure all core areas in the U.S. have secure water sources that provide for a quantity and quality of water sufficient to manage existing and newly created rail habitat.
 - As possible, provide protection for other habitat areas supporting breeding Yuma Ridgway's rails through

management plans associated with easements, mitigation associated with federal actions, habitat conservation plans, safe harbor agreements, the Partners for Fish and Wildlife Program, tribal cooperation, and other options.

4. Provide a mechanism for coordination and implementation of recovery actions.
 - Establish a recovery implementation team with responsibilities for implementing recovery activities, with emphasis on tasks relating to survey management, research, and development of partnerships.
 - Cooperate with partners in Mexico on issues related to long-term survival of Yuma Ridgway's rail.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Yuma Ridgway's rail, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 54,978 acres of modeled suitable habitat for Yuma Ridgway's rail in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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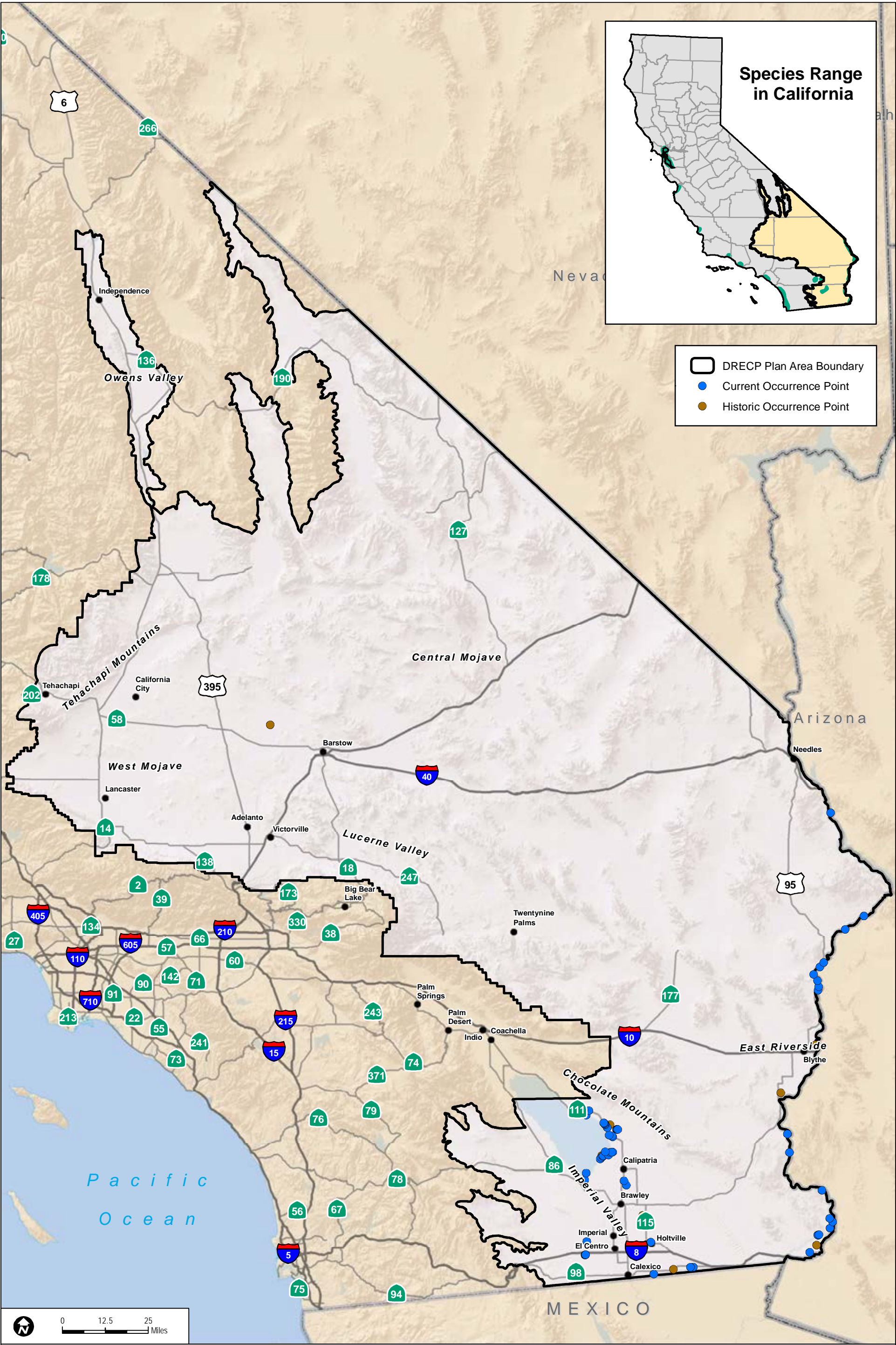
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BIRDS

Yuma Ridgway's Rail (*Rallus obsoletus yumanensis*)

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B14
Yuma Ridgway's Rail Occurrences in the Plan Area

Desert Pupfish (*Cyprinodon macularius*)

Legal Status

State: Endangered

Federal: Endangered

Critical Habitat: 51 FR
10842–10851

Recovery Planning: Desert Pupfish Recovery Plan (USFWS 1993)



Photo courtesy of Sharon Keeney, CDFW

Taxonomy

The desert pupfish complex was historically comprised of two subspecies, the nominal desert pupfish (*Cyprinodon macularius macularius*) and the Quitobaquito pupfish (*Cyprinodon macularius eremus*), and an undescribed species, the Monkey Spring pupfish (*Cyprinodon* sp.) (USFWS 1993). The subspecies are now recognized as three separate species (USFWS 2010): the desert pupfish (*C. macularius*), the Sonoyta (Quitobaquito) pupfish (*C. eremus*) (Echelle et al. 2000), and the undescribed Monkey Springs pupfish, which has since been described and renamed the Santa Cruz pupfish (*C. arcuatus*). Recent work (Echelle et al. 2007; Koike et al. 2008) and a summary by the U.S. Fish and Wildlife Service (USFWS 2010) provide the evidence that *C. macularius* and *C. eremus* are separate species. The Sonoyta pupfish persists in only two populations: one near the U.S.–Mexico border at Quitobaquito Springs in Organ Pipe Cactus National Monument in Arizona, and the other at Rio Sonoyta in Sonora, Mexico (USFWS 2010). The Santa Cruz pupfish occurred in the upper Santa Cruz River basin in southern Arizona and Northern Sonora, Mexico. It is now extinct due to habitat alteration and introduced fishes (Minckley et al. 2002). All other populations are referred to *C. macularius*. Descriptions of the species' physical characteristics can be found in USFWS (1993, 2010).

Distribution

General

The desert pupfish occurs in desert springs, marshes, and tributary streams of the lower Gila and Colorado River drainages in Arizona, California, and Mexico. Natural populations of desert pupfish also occur in the Salton Sea and associated irrigation drains and shoreline pools. It also formerly occurred in the slow-moving reaches of some large rivers, including the Colorado, Gila, and San Pedro.

Distribution and Occurrences within the Plan Area

Historical

Historically, desert pupfish occurred in the lower Colorado River in Arizona and California, from about Needles downstream to the Gulf of Mexico and onto its delta in Sonora and Baja (CVAG 2007). In California, pupfish inhabited springs, seeps, and slow-moving streams in the Salton Sink basin, and backwaters and sloughs along the Colorado River. Desert pupfish also occurred in the Gila River Basin in Arizona and Sonora, including the Gila, San Pedro, and Salt Rivers; Puerto Penasco, Sonora; and the Laguna Salada Basin of Baja California. The currently recognized historical range of desert pupfish has changed due to taxonomic changes. The recognition and naming of the Sonoyta (Quitobaquito) pupfish and Santa Cruz pupfish as separate species removed the Rio Sonoyta and Santa Cruz River basins from the previously known historical range of desert pupfish (USFWS 2010).

Recent

Because *C. eremus* occurs only in southern Arizona and Mexico (USFWS 2010) and *C. arcuatus* is now extinct, their distribution information is not discussed further; *C. macularius* is described within the Plan Area (see Figure SP-F01). USFWS (2010) describes that currently five natural populations persist in California, restricted to two streams tributary to, and many shoreline pools and irrigation drains of, the Salton Sea: San Felipe Creek/San Sebastian Marsh, Salt Creek (within the Dos Palmas Conservation Area of the Coachella Valley Multiple Species Habitat Conservation Plan [MSHCP; CVAG

2007]], Salton Sea, irrigation drains of the Salton Sea, and a wash near Hot Mineral Spa (a natural population added since the 1993 recovery plan). The desert pupfish population in Salt Creek is stable to increasing, and currently has few non-native species (Keeney 2010a, cited in USFWS 2010). San Felipe Creek also has a stable to increasing population. California Department of Fish and Wildlife (CDFW) surveys have found a persistent population of western mosquitofish (*Gambusia affinis*) in San Felipe Creek in recent years. In addition, there are a number of refuge or captive populations of desert pupfish in California at a variety of sites (USFWS 2010): Anza-Borrego State Park; Oasis Springs Ecological Reserve; Salton Sea State Recreation Area; Dos Palmas Reserve; Living Desert Museum; University of California, Riverside; and Borrego Springs High School. The Coachella Valley MSHCP (CVAG 2007) also describes a refuge population in the larger pools around the Thousand Palms oasis area where restoration is in progress. There are no pupfish currently present here, but there are plans to restock this site when restoration has been completed.

Natural History

Habitat Requirements

Found in water of desert springs, small streams, and marshes below 1,515 meters (5,000 feet) elevation (USFWS 1993), this species tolerates high salinities, high water temperatures, and low dissolved-oxygen concentrations. In the mid-2000s CDFW found desert pupfish in the Salton Sea at depths of 7 to 8 feet while conducting fish monitoring surveys. Pupfish typically prefer clear water, with either rooted or unattached aquatic plants, restricted surface flow, and sand-silt substrates (Black 1980; USFWS 1993). Pupfish use shallow water habitats extensively, often occupying such habitat at temperatures that are above the thermal optimum for invasive fishes. Pupfish do well if these habitats have little vegetation apart from mats of benthic algae over a fine-grained mineral or detrital substrate; they also utilize areas with aquatic or emergent vascular vegetation (ICF 2009). Desert pupfish in general are noted for their tolerance of environmental stress; they can tolerate dissolved-oxygen concentrations as low as 0.13 parts per million (Helfman et al. 1997). Their temperature tolerance ranges from a low of 4.4°C (Schoenherr 1990) to a high of 42.4°C (Carveth

et al. 2006). Their salinity tolerance ranges from 0 to 70 parts per thousand for eggs and adults (Barlow 1958; Schoenherr 1988) and up to 90 parts per thousand for larvae (Schoenherr 1988). Martin and Saiki (2005) found that desert pupfish abundance was higher when vegetative cover, pH, and salinity were high and when sediment factor and dissolved oxygen were low. They hypothesize that water quality extremes (especially high pH and salinity, and low dissolved oxygen) limit the occurrence of nonnative fishes.

Table 1. Habitat Associations for Desert Pupfish

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Water of desert springs, small streams, and marshes	Breeding/ foraging	Primary habitat	Clear water, with either rooted or unattached aquatic plants, restricted surface flow, and sand–silt substrates	Direct observational studies

Sources: Black 1980; USFWS 1993; Martin and Saiki 2005.

Foraging Requirements

Pupfish are opportunistic omnivores, thriving on a diet of algae, aquatic plants, detritus, and small invertebrates (Sutton 1999, citing Crear and Haydock 1971 and Naiman 1979). Adult foods include ostracods, copepods, and other crustaceans and insects; pile worms; mollusks; and bits of aquatic macrophytes torn from available tissues (USFWS 1993). Legner et al. (1975) found that desert pupfish were more effective than mosquitofish at controlling mosquito populations. Pupfish have also been known to eat their own eggs and young on occasion. Detritus or algae are often predominant in their diets (USFWS 1993). Pit digging, the active excavation of soft bottoms in search of food, is a pupfish behavior described by Minckley and Arnold (1969); these pits are defended when occupied. Foraging is typically a daytime activity, and fish may move in response to daily warming from shallower water during morning to feed in deeper places later in the day (USFWS 1993).

Reproduction

Desert pupfish may become sexually mature as early as 6 weeks of age at 1.5 centimeters in length under conditions of abundant food and suitable temperature. Desert pupfish typically live for a year, but may live as long as 2 to 3 years. Although they may breed during their first summer, most do not breed until their second summer, when their length may have reached a maximum of 7.5 centimeters (Moyle 2002). In favorable conditions a pair of pupfish can produce 800 eggs in a season (ICF 2009). Eggs appear to be randomly deposited within the male territory. Although males actively patrol and defend individual territories, there is no directed parental care (USFWS 1993).

Table 2. Key Seasonal Periods for Desert Pupfish

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			X	X	X	X	X	X	X			

Source: USFWS 1993.

Spatial Behavior

McMahon and Tash (1988) found that when desert pupfish occupied open pools, 84% of the total number produced emigrated. They found that when pupfish were prevented from emigrating, pupfish exhibited symptoms of overpopulation. Characteristics of overpopulation were not apparent in pupfish occupying open pools. Seasonal temperatures influenced the timing and magnitude of emigration. In summary, pupfish may regulate their populations via emigration.

Many of the locations where they are currently found are isolated from other populations. However, complete isolation mainly has been an issue in artificial populations, although even in these populations “complete isolation” no longer occurs given CDFW’s recent inoculation of refuges with wild fish. Most natural populations have some connection to other populations occasionally (e.g., via flash flood), although these opportunities for mixing are brief and infrequent. This may become more of an issue given the uncertainty of the Salton Sea.

Desert pupfish congregate in the summer where adult females swim in loose schools and leave the school when attracted by a territorial male to spawn. Pupfish movement between the Salton Sea and nearby drains has been observed (Sutton 1999). Sutton (2002) describes desert pupfish summer movement between a drain (although not connected directly to the Salton Sea) and a shoreline pool, as well as movement of approximately 0.5 kilometer (0.3 mile) from Salt Creek to a downstream shoreline pool (although not connected to the Salton Sea). Sutton (2002) hypothesizes that movements from Salt Creek to the shoreline pool were due to water level drops. The technique used by Sutton (2002) for tracking desert pupfish holds promise for further desert pupfish movement studies.

Table 3. Spatial Behavior by Desert Pupfish

Type	Distance/Area	Location of Study	Citation
Breeding territory	Normally defends 1 to 2 square meters but as large as 5 to 6 square meters	Not disclosed	Moyle 1976

Ecological Relationships

The desert pupfish were once found in varying water bodies from cienegas and springs to shallow streams and margins of larger bodies of water where they preferred shallow, slower-moving water with soft substrates and clear water (USFWS 1993). Over the last century, land use activities such as groundwater pumping, dewatering, water diversion, and drain maintenance have altered the water levels, resulting in habitat loss for desert pupfish. Channel erosion can increase the sediment in the water, reducing its suitability for the pupfish; water impoundment creates deeper ponds that increase occupation by non-native aquatic species; and grazing practices reduce vegetative cover, increase sedimentation, and trample habitat (USFWS 1993). Off-road vehicle use can be problematic in some areas, and currently is more of an issue than is grazing.

Currently, the major threat to the species is the presence of exotic aquatic species, particularly tilapia (*Tilapia* spp.), sailfin molly (*Poecilia latipinna*), western mosquitofish, several snail species, and crayfish

(*Procambarus clarkii*). These and other introduced fish species primarily affect pupfish populations through predation, competition, and behavioral interference (CVAG 2007). Introduced fishes (and other aquatic organisms) can affect pupfish populations via other means as well, such as disease and habitat displacement. Additionally, in a few areas, such as San Felipe Creek and Salt Creek, where non-native fishes are relatively few (at least currently), the most serious threat may be the abundance of tamarisk/salt cedar (*Tamarix* spp.).

The desert pupfish appears to go through cycles of expansion and contraction in response to natural weather patterns (51 FR 10842–10851; USFWS 1993; Weedman and Young 1997, cited in USFWS 2010). In very wet years, populations can rapidly expand into new habitats (Hendrickson and Varela-Romero 1989, cited in USFWS 2010). In historical times, this scenario would have led to panmixia among populations over a very large geographic area (USFWS 1993).

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)

State: Same as above

Within Plan Area: Same as above

In its 5-year review, USFWS (2010) concluded that threats to the species and their overall level of intensity remain similar to when the species was originally given a recovery priority number of 2C. Priority number 2C is indicative of a high degree of threat, a high potential for recovery, and taxonomic classification as a species.

Threats and Environmental Stressors

USFWS (2010) summarizes the threats to desert pupfish survival. These include threats relating to destruction or curtailment of habitat or range (USFWS Factor A), including loss and degradation of suitable habitat through groundwater pumping or water diversion; contamination from agricultural return flows, as well as other contaminants; and physical changes to water properties involving suitable water quality. There is no new information to suggest that overutilization for commercial, recreational, scientific, or educational purposes (USFWS Factor B) are threats. The effect of disease or predation (USFWS Factor C) is a potential threat to desert pupfish.

Currently, the specific effects to individual desert pupfish or populations from disease or parasites are unknown. Predators and competitors of the desert pupfish include tilapia, sailfin mollies, shortfin mollies (*Poecilia mexicana*), mosquitofish, porthole livebearers (*Poeciliopsis gracilis*), and several members of the families *Centrarchidae*, *Ictaluridae*, and *Cyprinidae*, as well as melanias (*Melanoides tuberculata* and *Tarebia granifera*), crayfish, Rio Grande leopard frog (*Lithobates berlandieri*), and bullfrog (*Rana catesbeiana*) (51 FR 10842–10851; Black 1980; ICF 2009). Invasive snails (melanias) consume the algal mats that form the pupfish's principal food source (ICF 2009). They also may cause disease. For example, red-rim melania (*Melanoides tuberculatus*) is a host of parasites, including gill trematode. Known fish hosts of the gill trematode include Comanche Springs pupfish (*Cyprinodon elegans*). Juvenile tilapias compete with desert pupfish for many of the same food items (Matsui 1981); and crayfish, frogs and adult tilapia prey on fish and fish eggs (51 FR 10842–10851; ICF 2009; Matsui 1981). Crayfish were thought to be responsible for elimination of the Owens pupfish, *C. radiosus*, from a refuge in Warm Springs near Big Pine, California (Black 1980). Additionally non-native crayfishes are well known to negatively affect water quality and severely reduce, if not eliminate, algae that is favored by pupfish. These and other introduced aquatic species affect pupfish populations through predation, competition, and behavioral interference. Inadequacy of existing regulatory mechanisms (USFWS Factor D) is a potential threat to desert pupfish. Regulatory mechanisms exist in much the same state as at the time of listing, though the application of recent case law may result in reduced consideration of impacts to isolated waters containing desert pupfish (USFWS 2010). Finally, other natural or manmade factors affecting the continued existence of desert pupfish (USFWS Factor E) have been noted as a threat for desert pupfish (USFWS 1993). The only new threat identified is endocrine disruptors noted in the Salton Sea irrigation drains (USFWS 2010).

Conservation and Management Activities

The Coachella Valley MSHCP (CVAG 2007) lists some conservation and management actions that would benefit pupfish:

1. Complete hydrologic studies for the Salt Creek area to determine if the water sources for Salt Creek are adequately protected or if additional water sources may be needed and are available.¹
2. Ensure persistence of pupfish populations in agricultural drains by managing agricultural drain maintenance and water supply. Monitoring will include surveys for pupfish presence in the agricultural drains along with regular sampling of flow, water depth, and selenium concentrations
3. Control and manage exotic or invasive species in pupfish habitat, if monitoring identifies this as a threat. Control efforts should address nonnative fish, bullfrogs, and other invasive species. The presence and potential impacts of Asian tapeworm, a potential pupfish parasite, shall also be addressed.
 - a. Remove tamarisk (salt cedar) where it is affecting the amount of water available to pupfish.
4. Maintain water levels, water quality, and proper functioning condition of ponds, springs, and drains, to the extent these activities are under Plan authority, which will include reevaluating the feasibility of available technologies to reduce selenium concentrations.
5. Restore and enhance degraded habitat as necessary according to monitoring results.
6. Conduct experiments on the timing and mechanics of drain cleaning that would minimize impacts to desert pupfish.
7. Estimate distribution and/or population size of desert pupfish.
8. Survey contaminant levels in the water and in pupfish.

¹ San Felipe Creek and associated wetlands are not within the Coachella Valley MSHCP area, but complete hydrologic studies are needed for this system as well. This will be particularly important given potential impacts of climate change.

USFWS (2010) also lists some general future conservation and management activities:

- A specific standardized genetic protocol should be developed, using work by Echelle et al. (2007), as a template for management of *C. macularius* refuge populations. CDFW is currently working on this issue as part of the Desert Pupfish Refuge Management Plan being developed to provide guidance for the management of pupfish refuges (artificial habitats). Their recommendations include establishing large primary refuge populations, with each one representing the groups of wild *C. macularius*. They also recommend that secondary refuges representing each of the wild source regions be established.
- A recovery plan amendment or revision should be made based on recommendations by Loftis et al. (2009) that delineate a different set of management units in the Salton Sea than is recognized in the existing recovery plan and to reflect the changed taxonomy.
- Conservation at wild sites should be given the highest priority.
- A Safe Harbor Agreement or similar tool for the desert pupfish in California should be pursued.

Additionally, another desired study is determining the tolerance of pupfish eggs to desiccation; this study is currently being planned and is expected to occur soon.

Data Characterization

Loftis et al. (2009) assessed the mitochondrial DNA (mtDNA) results from the 1997 and 1998 surveys by Echelle et al. (2000) and used data from 10 microsatellite DNA loci to describe the genetic structure of the two extant species (*C. macularius* and *C. eremus*). According to Loftis et al., this data showed that there “was evidence ($R_{ST} > F_{ST}$) that the two extant populations of *C. eremus* have been isolated sufficiently long for mutation to contribute significantly to genetic divergence, whereas divergence among the nine assayed populations of *C. macularius* could be attributed to genetic drift alone.” The assessment suggests that based on variability among the mtDNA, there are two populations of *C. eremus* and five groups of populations of *C.*

macularius that should be managed as units for conservation genetics management of the two species.

The distribution of the species and principal threats to its continued existence are sufficiently well known to allow coverage of this species in the Desert Renewable Energy Conservation Plan.

Management and Monitoring Considerations

As summarized above, the Coachella Valley MSHCP (CVAG 2007) lists some specific conservation and management actions for the Plan Area that would benefit pupfish. In addition, invasive species management options for the Dos Palmas Area of Critical Environmental Concern have been prepared (ICF 2009) and cover threats to the desert pupfish. Within that document, specific management actions that may be used to eliminate non-native aquatic species or create predator-free environments are evaluated; these include water management that alternately inundates and desiccates habitat, creation of channel habitat, creation of shallow-water habitat, removal and/or burning of emergent aquatic habitat, and invasive aquatic species trapping. As mentioned previously, CDFW is preparing the Desert Pupfish Refuge Management Plan, which will address specific management issues including control of aquatic fauna and flora, genetic protocols for monitoring of pupfish, management recommendations for each refuge, pupfish population monitoring, and other topics. The Desert Pupfish Recovery Plan (USFWS 1993) emphasizes securing extant wild populations of desert pupfish to preserve original genetic material, and creating a second and third tier of populations from these existing wild populations using a genetic exchange protocol that would be created to mimic desert pupfish evolution. Refuge population or new habitat may not be difficult to create as is evidenced by the shallow-water habitat that was constructed near the Alamo River, which was designed to exclude fish, but desert pupfish got into the ponds and flourished (Roberts 2010, as cited in USFWS 2010; Saiki et al. 2011). However, habitat may be difficult to maintain in terms of costs. Bureau of Reclamation spent three million dollars constructing, operating and maintaining this habitat before running out of funding.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for desert pupfish, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 8,155 acres of modeled suitable habitat for desert pupfish in the Plan Area. A figure showing the modeled suitable habitat in the Plan Area are included in Appendix C.

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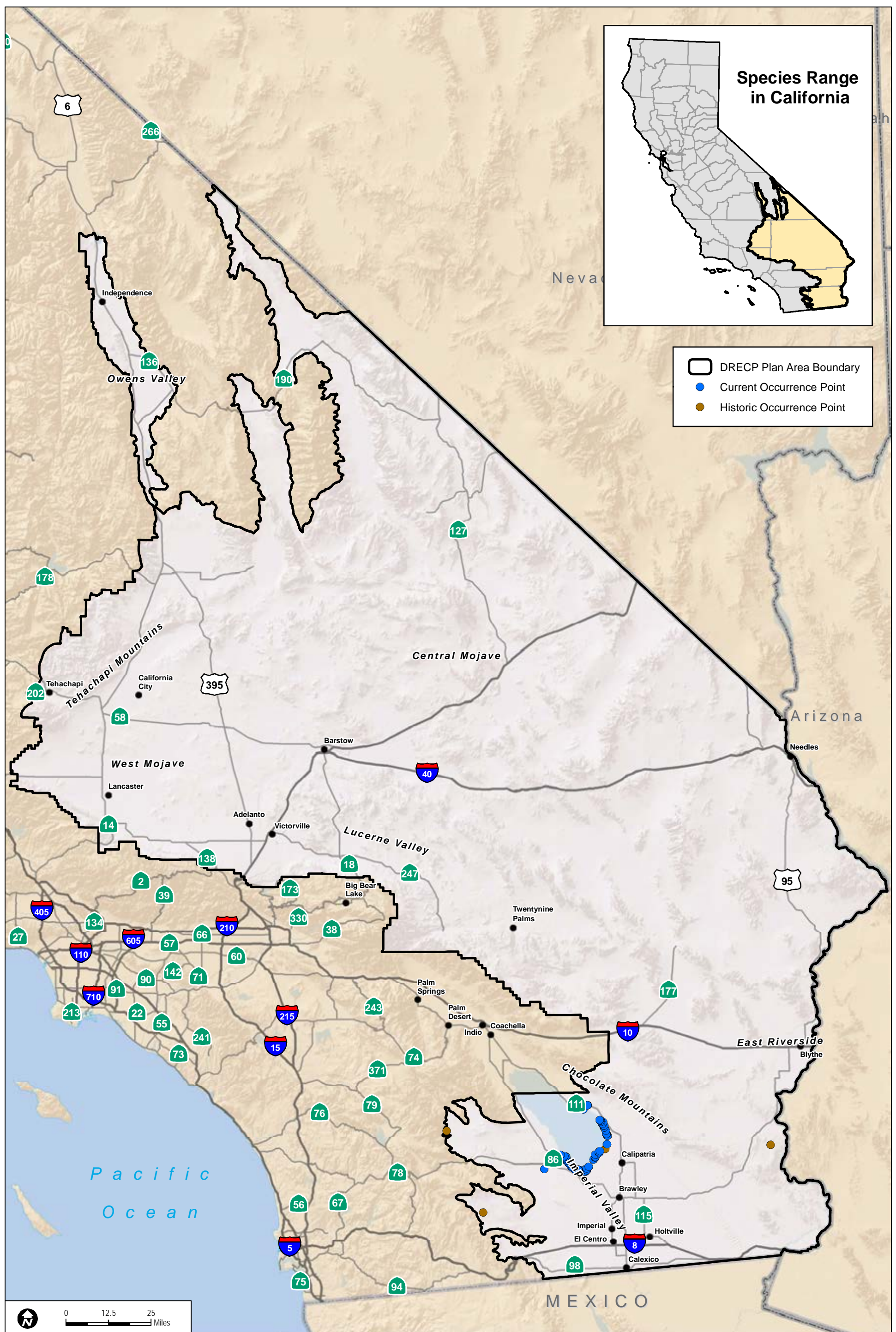
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Mohave Tui Chub

(*Siphateles bicolor mohavensis*)

Legal Status

State: Endangered, Fully Protected

Federal: Endangered

Critical Habitat: N/A

Recovery Planning: Recovery Plan for the Mohave Tui Chub, *Gila bicolor mohavensis* (USFWS 1984)

Notes: California Department of Fish and Wildlife (CDFW) has adopted the genus *Siphateles* for the species, which was previously classified under the genus *Gila*.

Taxonomy

The Mohave tui chub (*Siphateles bicolor mohavensis*) is recognized as the only fish native to the Mojave River basin in San Bernardino County. It is a member of the minnow family (Cyprinidae). It was originally identified as *Algansea formosa* in 1857 by Girard, but in 1918 Snyder described it as a new species, *Siphateles mohavensis* (as cited in USFWS 1984). Miller (1961) and Bailey and Uyeno (1964) relegated the subgenus *Siphateles* to the genus *Gila*, and in 1973 Miller reclassified the Mohave tui chub to the subspecies *G. b. mohavensis* (as cited in USFWS 1984). Simons and Mayden (1998) published a paper addressing the classification of the North America genera of Cyprinidae and, based on ribonucleic acid (RNA) sequences, restored *Siphateles* from a subgenus to a full genus. The CDFW currently includes the species under the genus *Siphateles* (CDFW 2013), and the U.S. Fish and Wildlife Service (USFWS) intends to propose amending Part 17, Subchapter B of Chapter I, Title 50, of the Code of Federal Regulations to reflect the taxonomic change from *G. b. mohavensis* to *Siphateles bicolor mohavensis* (USFWS 2009). This taxonomic change will not affect its federal listing status. A physical description of the species can be found in the 5-Year Review (USFWS 2009).

The Mohave tui chub has a distinct lineage and is a separate subspecies from its closest relative, the Lahontan Lake and Lahontan creek tui chubs (*Siphateles bicolor pectinifer* and *Siphateles bicolor*

obesa, respectively). Mohave tui chub is least similar genetically to arroyo chub (*Gila orcutti*) (USFWS 2009).

Distribution

General

Historically, the Mohave tui chub is believed to have occurred throughout the Mojave River drainage (Miller 1946, cited in USFWS 1984). According to the Recovery Plan for the Mohave Tui Chub, *Gila bicolor mohavensis* (Recovery Plan) (USFWS 2009), the Mojave River drainage in the Mojave Desert originally consisted of the Mojave, Little Mojave, and Manix lakes; during the Pleistocene age, these lakes were connected through channels, and Mohave tui chubs were probably found throughout the drainage (Figure 2; USFWS 1984). As the climate became drier and the lakes receded, the Mohave tui chub was restricted to the Mojave River. During the 1930s, arroyo chubs were introduced into the Mojave River and likely hybridized with the Mohave tui chub, thus eliminating the genetically pure Mohave tui chub within the Mojave River (USFWS 1984). A small population of genetically pure Mohave tui chub persisted in isolated ponds near the terminus of the Mojave River at Soda Springs. Four populations of the Mohave tui chub have also been successfully introduced at the Lark Seep complex at China Lake Naval Weapons Station, Camp Cady Wildlife Area (USFWS 2009), the Lewis Center in Apple Valley, and Morning Star Mine at Mojave National Preserve. All of these populations are located within the Plan Area.

Distribution and Occurrences within the Plan Area

Historical

As described above, the Mohave tui chub was historically found within the Mojave River basin as the only native fish within this system. By 1970, the genetically pure Mohave tui chub had been eliminated from the Mojave River due to several factors, including hybridization; introduction of other non-native, competitive, and predatory aquatic species to its historical habitat (e.g., bass [*Micropterus* spp.], catfish [*Ictalurus* spp.], trout [*Oncorhynchus* spp.], bullfrog [*Rana catesbeiana*], and crayfish [*Procambarus clarkii*] [Miller 1969]); habitat alteration; water diversions; and pollution

(USFWS 2009). At the time of listing in 1970, four populations were known to exist; three were located in San Bernardino County at Piute Creek, Two Hole Spring, and Soda Springs; and one was in Paradise Spa, Nevada (USFWS 2009). There are nine historical (i.e., pre-1990) records in the Plan Area contained in the California Natural Diversity Database, occurring in the eastern end of Mojave National Preserve and along the northern flank of the San Bernardino Mountains (Figure SP-F02) (CDFW 2013; Dudek 2013).

Recent

A population was established in 1978 at the Desert Research Station near Hinkley, California; however, in 1992 the pond dried up and the population was extirpated. As of 2011, there were five populations of genetically pure Mohave tui chubs: Soda Springs and Morning Star Mine at Mojave National Preserve, Lark Seep at China Lake Naval Air Weapons Station, Camp Cady Wildlife Area, and the Lewis Center in Apple Valley (Figure SP-F02). All of these locations are within the Plan Area. The Camp Cady Wildlife Area is managed by CDFW; Soda Springs Mojave National Preserve and Morning Star Mine are managed by the National Park Service; and the Lark Seep complex is located on a naval base managed by the Department of Defense.

Natural History

Habitat Requirements

Historically, within the Mojave River, the Mohave tui chub was associated with deep pools and sloughs of the river and was not found very far into small tributaries (USFWS 1984). Although the Mohave tui chub does not currently occupy the Mojave River, a few perennial stretches of the river remain that could support a fishery. The habitat requirements for this species include configuration, ecology, and water quality (Archbold 1996, as cited in USFWS 2009). The configuration of a lacustrine pond or pool should include a minimum water depth of 4 feet with some freshwater flow for a mineralized and alkaline environment (USFWS 2009; NatureServe 2011). The pools or ponds should include some aquatic plants (e.g., *Ruppia maritima*, *Typha* spp., and *Juncus* spp.), which provide habitat for aquatic invertebrates consumed by Mohave tui chub and a substrate for egg attachment (USFWS 2009). Aquatic ditchgrass

(*Ruppia maritima*) appears to be the preferred vegetation for egg attachment and thermal refuge in summer months (USFWS 1984). In addition, the Mohave tui chub is sensitive to predation from other fish species, and pools should be relatively free of arroyo chubs and other non-native aquatic wildlife species (USFWS 2009). Finally, to be suitable for Mohave tui chub, the water should have water quality parameters within the tolerable range for this species and be free of toxic substances or the threat of toxic substance spills (USFWS 2009). Water quality parameters include a temperature range from 37° Fahrenheit (F) to 97°F, dissolved oxygen at greater than 2 parts per million, a salinity of 40 to 323 milliosmols per liter, and a pH of up to 9 with 10 being tolerable for a short period of time (Feldmeth et al. 1985; Archbold 1996; and McClanahan et al. 1986, cited in USFWS 2009).

The current populations are located in primarily man-made or man-supported habitats. The population in Lark Seep is in a perennial body of water that is fed from the wastewater treatment facility in Ridgecrest, California. The population at Camp Cady is located in a man-made, lined pond that receives water from a pump. The populations at Soda Springs occur in two bodies of water, one is a man-made pond that receives water from a pump, and the other is an isolated spring on the edge of Soda Lake (USFWS 2009). The population at the Lewis Center is in two small man-made ponds with water supplied from a pump, and at Morning Star Mine, the population is in a man-made pond created by a perched aquifer. Table 1 lists primary habitat associations and parameters for Mohave tui chub.

Table 1. Habitat Associations for Mohave Tui Chub

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Lacustrine ponds/pools	All life history phases	Primary	Minimum depth of 4 feet and water quality limitations	USFWS 1984, 2009

Foraging Requirements

Not much is known about the specific diet of the Mohave tui chub. They forage on a variety of aquatic invertebrates, including plankton

and insect larvae, small fish and organic detritus (Archdeacon 2007, cited in USFWS 2009; NatureServe 2011). Ponds and pools that have aquatic vegetation provide habitat for these food sources, as discussed previously under Habitat Requirements (USFWS 2009).

Reproduction

Mohave tui chubs spawn after 1 year of age (USFWS 1984). Spawning begins during the spring in March and April when water temperatures are warm enough (64°F) (Vickers 1973, cited in USFWS 1984). Spawning may occur in the fall as well. Egg masses are laid in vegetation where they become attached after fertilization. The eggs are approximately 0.04 inch in diameter and hatch after approximately 6 to 8 days when water temperatures are between 64° F and 68° F (USFWS 1984).

Spatial Activity

Currently, the populations of Mohave tui chub are restricted to ponds and man-made channels where they do not have any connection to other populations. Past efforts to introduce or transplant additional populations generally have not been successful (USFWS 2009) with the exception of their current locations in Kern and San Bernardino Counties, California.

Ecological Relationships

The Mohave tui chub originated from the Mojave River basin where it was adapted to the perennial deep pools and slough-like areas of the Mojave River and an absence of aquatic predators. Several factors contributed to its decline and current status as a federal and state listed species. The introduction of arroyo chub into the Mojave River in the 1930s resulted in likely hybridization and elimination of genetically pure Mohave tui chub species. The arroyo chub was also a source of competition for food.

Flooding, changes in water quality, and the introduction of non-native plant and wildlife species have also affected this species (USFWS 1984). Flooding in the Mojave River in 1938 enabled arroyo chubs to disperse further throughout the Mojave River system, and because of their adaptation to waters with greater velocities, the arroyo chub

was successful at surviving these floods. Mohave tui chubs, on the other hand, are adapted to lacustrine conditions and are not able to persist in conditions with high-velocity flow and warmer shallow channels (USFWS 2009). These adaptive differences have contributed to replacement of Mohave tui chub by arroyo chub (Castleberry and Cech 1986). In addition, changes in water quality and quantity have resulted in the loss of subpopulations at East Pond (Camp Cady) and Three Bats Pond (Soda Springs) (USFWS 2009). The introduction of non-native plants and aquatic and amphibious species into the Mojave River system has resulted in modification of the species' habitat. Predation by introduced aquatic species (e.g., bass [*Micropterus* spp.], trout [*Oncorhynchus* spp.], catfish [*Ictalurus* spp.], mosquitofish [*Gambusia affinis*], and bullfrogs [*Lithobates catesbeianus*]) contributed to the extirpation of the Mohave tui chub in the Mojave River (USFWS 2009). The establishment of salt cedar (*Tamarix* sp.), has altered water flow and geomorphology of the Mojave River system (Lovich 2006).

A study conducted at Fort Soda in 1981–1982 found that Mohave tui chub populations increased two to three times during the spring and summer months, and then decreased during the fall and winter months (Taylor 1982). A study examining the growth and population structure of the Mohave tui chub at a research station northwest of Barstow in the 1980s found that the population was highest in late summer and lowest in late winter (Havelka et al. 1982). Tui chubs gained weight in May, but lost up to 35% of their body weight from June to October before gaining weight again in November. This may be the result of higher metabolic rates during the summer coupled with a possible reduction in planktonic biomass (Havelka et al. 1982).

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)

State: Same as above

Within Plan Area: Same as above

As described previously under Distribution, Mohave tui chub is only present at five locations, and remains extirpated from its historic habitat in the Mojave River. As concluded in the 2009 5-Year Review for the species, the Mohave tui chub “still meets the definition of

endangered in the Act for the following reasons: (1) there are fewer populations of this subspecies now than at the time of listing; (2) the rare nature of this subspecies increases the risk of local extirpations from stochastic events; (3) all populations of the Mohave tui chub are threatened by one or more of the threats described in the Recovery Plan that contributed to its endangered status including habitat loss and alteration, predation from non-native species, with the additional, newly identified threats of parasitism, genetic drift, and extirpation from stochastic events; (4) the lack of consistent and reliable management and monitoring activities for these populations, which makes it difficult to identify and determine the magnitude and imminence of current threats, and therefore, to ensure that the threats will be identified in time and ameliorated; and (5) the failure to meet any of the downlisting or delisting criteria in the Recovery Plan” (USFWS 2009).

Threats and Environmental Stressors

The American Fisheries Society publication of its endangered, threatened, or of special concern fishes of North America identified two main threats to Mohave tui chub: 1) the present threatened destruction, modification, or curtailment of its habitat or range; and 2) other natural or man-made factors affecting its continued existence (hybridization, introduction of non-native or transplanted species, predation, or competition) (Williams et al. 1989, cited in USFWS 2009).

The Mohave tui chub is already extirpated from its historical distribution in the Mojave River. As one of the criteria for delisting the Mohave tui chub, the Recovery Plan includes the return of the Mohave tui chub into its historical range in the Mojave River. Over the years, the aquifer of the Mojave River has been overdrafted, resulting in the loss of aquatic habitat. Many of the areas within the river are now shallow and lack the lacustrine conditions once characteristic of portions of the Mojave River drainage, thus reducing the suitable habitat available for Mohave tui chub reintroduction.

A parasitic Asian tapeworm was found in Lake Tuendae (Soda Springs), and it initially had a deleterious effect on the population there. It was found to contribute to a reduced growth rate of Mohave tui chub in captivity, but not the survival rate (Archdeacon 2007). Research on Asian tapeworm parasitism has shown no long-term

debilitating impacts on Mohave tui chub populations (Archdeacon 2007, cited in USFWS 2009).

Non-native species, such as bullfrogs and sport fish (e.g., bass and catfish), were introduced into the river. Predation on Mohave tui chub from these species contributed to its extirpation within the Mojave River (Williams et al. 1989, cited in USFWS 2009). Mosquitofish were found in Lake Tuendae (Soda Springs) in 2001 and were found to reduce the survival rate of the chubs when no cover is provided in the environment (Archdeacon 2007). They also compete for food and other resources, which may pose a threat to the Mohave tui chub.

Other threats to the Mohave tui chub include regulatory mechanisms. For example, USFWS (2009) states that the military installations do not obtain incidental take permits under the California Endangered Species Act; however, China Lake Naval Air Weapons Station implements Section 7(a)(1) of the federal Endangered Species Act, which requires federal agencies to utilize their authorities in the furtherance of the purposes of the act by carrying out programs for the conservation of federally endangered and threatened species. It should be noted that at the time of the 5-Year Review, the only proposed activities that would result in the take of Mohave tui chub were for research permits, which is purposeful take (USFWS 2009).

Conservation and Management Activities

The USFWS and cooperating agencies have proposed establishing additional populations of Mohave tui chub in the Mojave River watershed and the California portion of the Mojave Desert in order to contribute to the conservation of the Mohave tui chub (USFWS 2011). An environmental assessment has been completed to analyze the locations where these populations could be established (USFWS 2011).

Because all of the current populations of Mohave tui chub occur in man-made or man-supported environments, ongoing conservation and management activities are required. To ensure the long-term sustainability of the Mohave tui chub, the 5-Year Review indicates that habitat management, ecosystem restoration, monitoring, and adaptive management are needed (USFWS 2009).

All of the current populations require regular control of cattails (*Typha* spp.) in ponds to maintain open water environments and suitable water conditions. Other specific management considerations include the Asian tapeworm, mosquitofish, habitat loss and degradation, water quality and supply, and genetic drift (USFWS 2009). Genetic drift can result in a loss of alleles (i.e., genetic variation) at small, isolated populations and can result in increased risk of extirpation. Recent data indicate that populations at MC Spring (at Soda Springs) and Camp Cady have recently shown a loss of genetic diversity (S. Parmenter, pers. comm. 2007, cited in USFWS 2009).

Data Characterization

To better manage and recover the species, the 5-Year Review (USFWS 2009) suggests identifying the extent and magnitude of bird predation, determining spawning requirements and early life history, determining physiological tolerances of Mohave tui chubs and arroyo chubs to water quality parameters, and identifying genetic issues, such as founder effect and possible hybridization with arroyo chubs.

Recent genetic analysis indicates that all existing populations of Mohave tui chubs are genetically pure; they do not show genetic evidence of hybridization with arroyo chubs. While the Mohave tui chub populations at Lark Seep and the Lake Tuendae subpopulation of Soda Springs are heterogeneous, genetic drift, or a loss of alleles, has occurred at the MC Spring subpopulation of Soda Springs and Camp Cady (USFWS 2009).

Hybridization between Mohave tui chub and the Los Angeles Basin endemic arroyo chub was identified as a primary threat to the Mohave tui chub after arroyo chubs were introduced to the Mojave River in the 1930s. However, hybridization between these two fish has never been studied and documented. Mojave National Preserve has initiated research on the ability of these two fish to hybridize (USFWS 2009).

Management and Monitoring Considerations

Management and monitoring considerations are addressed in the Recovery Plan (USFWS 1984) and 5-Year Review (USFWS 2009) as actions necessary to downlist and delist the species. The overall

objective of the Recovery Plan for delisting is to reintroduce a viable, sustainable population of Mohave tui chub into a majority of its historic habitat in the Mojave River (USFWS 1984). To achieve this objective, several management activities must occur, including management of introduced aquatic predators, hybridization with arroyo chub, water supply, water quality, and suitable habitat (e.g., deep, cool pools and sloughs).

In the interim, the Recovery Plan identified objectives to downlist the species from endangered to threatened. These objectives include establishing six populations of at least 500 Mohave tui chub in each population. Currently, there are only three populations that meet this criterion. Portions of the Mojave River that have been identified for additional potential reintroduction include the Mojave Narrows Regional Park area in Victorville, Camp Cady, portions of Afton Canyon, and an area downstream from the Victor Valley wastewater treatment facility in Oro Grande (USFWS 2009). However, it is likely that habitat management of these areas would be required because many of them have shallow flows rather than the preferred habitats of pools and sloughs.

Because all of these areas identified for potential reintroduction are located within the Plan Area, there should be careful consideration of future activities that could affect these areas.

Furthermore, the American Fisheries Society has published guidelines for introductions of threatened and endangered species that could be applied to Mohave tui chub (Williams et al. 2011). The guidelines recommend restricting introductions to sites within the native or historic habitat, sites that are protected, sites where the potential for dispersal has been determined acceptable, sites that fulfill the species' life history requirements, and sites that contain sufficient habitat to support a viable population. In addition, introduction sites should be avoided where endangered or threatened fish could hybridize with other taxa or where other rare or endemic taxa could be adversely affected. The introduction stock should be from an appropriate source, should be examined for taxonomic status and presence of undesirable pathogens, should be of sufficient number and character, should be carefully and quickly transported, should be introduced under favorable conditions, and the translocation procedures should be documented. After translocation, the American Fisheries Society

recommends systematic monitoring of introduced populations, which involves restocking if necessary, determining the cause of any failures, and documenting findings and conclusions reached during the post-introduction (Williams et al. 2011).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mohave tui chub, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 360 acres of modeled suitable habitat for Mohave tui chub in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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Owens Pupfish

(*Cyprinodon radiosus*)

Legal Status

State: Endangered, Fully Protected

Federal: Endangered

Critical Habitat: N/A

Recovery Planning: *Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties, California* (USFWS 1998)

Notes: Species was federally listed endangered on March 11, 1967. It was listed as endangered in California in 1971 (USFWS 2009).

Taxonomy

The first taxonomic description of Owens pupfish (*Cyprinodon radiosus*) was in 1948 by Miller, but occurrence locations along with relative abundance observations of Owens pupfish were noted as early as 1859 by explorers and scientists (USFWS 2009). Owens pupfish is in the killifish family (Cyprinodontidae) and is one of five pupfish species native to California (BLM 2011). The desert pupfish (*Cyprinodon macularius*), which occurs in the lower Colorado River system, is the closest relative of the Owens pupfish (USFWS 1998). Though Owens pupfish is a member of the *C. nevadensis* complex, a group of four species in two closed basins of the Death Valley System in California and Nevada (Owens River Valley and Ash Meadows–Death Valley), it appears to be more closely related to *C. macularius* than to the Ash Meadows–Death Valley members of the complex (*C. diabolis*, *C. nevadensis*, and *C. salinus*). Apparently, *C. radiosus* and *C. macularius* share both a general morphological similarity and an ancestral mitochondrial deoxyribonucleic acid (DNA) that separates them from the Ash Meadows–Death Valley pupfishes (Echelle and Dowling 1992). Descriptions of the species' physical characteristics can be found in the U.S. Fish and Wildlife Service (USFWS) 5-Year Review (2009).

Distribution

General

The Owens pupfish is restricted to the Owens Valley portion of the Owens River in Mono and Inyo counties, California (Figure SP-F03). Based on historical observations, Owens pupfish is believed to have occupied all of the Owens River and possibly the Owens River Delta at Owens Lake. Currently, it occurs at Fish Slough, Mule Springs, Well 368, and Warm Springs (USFWS 2009). Eight of the 17 California Natural Diversity Database (CNDDB) occurrences are within the Plan Area, while the remaining occurrences are farther north and east of the Plan Area (CDFW 2013).

Distribution and Occurrences within the Plan Area

Historical

Five of the eight occurrences in the Plan Area were last documented prior to 1990 (Figure SP-F03). All of these are found within the Owens Valley in Inyo County and have possibly been extirpated (CDFW 2013; Dudek 2013).

Recent

Three recent occurrences (i.e., since 1990) of Owens pupfish occur in the Plan Area. One occurrence is at Well 368, located 0.2 mile west of the Owens River and 2.5 miles south of Mazourka Canyon Road. Last observed in 1999, this occurrence is presumed extant. In 1988, pupfish from Warm Springs were introduced into the ponds at this location, and both adults and juveniles were abundant throughout the North Fork Area in 1999. It is owned by the Los Angeles Department of Water and Power (CDFW 2013; Dudek 2013).

Natural History

Habitat Requirements

Owens pupfish occurs in shallow water habitats in the Owens Valley (CDFW 2013). It will occupy most aquatic habitat where water is relatively warm and food is plentiful (USFWS 2009). However, it

prefers warm, clear, shallow water, free of exotic fishes, and requires areas of soft substrate for spawning (CDFW 2013; USFWS 2009). In addition, Owens pupfish habitat differs from the habitat of other pupfish. Specifically, aquatic habitats associated with the Owens River are typically colder, frequently covered by ice during winter, and lower in conductivity and salinity than habitats occupied by other pupfish species (USFWS 2009).

All life stages may be found in the various microhabitats available with little apparent documented preference. However, adults frequently occupy deeper water than juveniles. Male pupfish are territorial and defend areas of substrate from competing males. Females occupy habitats along the margins of these territories (USFWS 2009). Table 1 lists primary habitat associations and parameters for Owens pupfish.

Table 1. Habitat Associations for Owens Pupfish

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Warm, clear, shallow aquatic habitat	Primary	Soft substrates required for spawning	CDFW 2013; USFWS 2009

Notes: Species only occurs in the Owens River.

Foraging Requirements

Owens pupfish are opportunistic omnivores and consume a variety of plant and animal foods. Their diet changes seasonally and generally includes whatever invertebrates and plants are most abundant at that time (USFWS 1998). However, they primarily feed on aquatic insects and are an effective biological control agent for mosquitos (USFWS 2009; USFWS 1998). They do not prey on other fishes (USFWS 1998).

Reproduction

Owens pupfish breed from April through October (BLM 2011). Females spawn over soft substrates in spring and summer when water temperatures are near 14°Celsius (C) (57°Fahrenheit [F]) (USFWS 1998). They may spawn up to 200 times per day, laying one or two eggs at a time (USFWS 2009). Males are very aggressive during

the breeding season as they protect their breeding territory (BLM 2011). Incubation lasts for approximately 6 days before hatching in water that ranges in temperature from 75°F to 81°F. On average, 95% of spawned eggs are fertilized. Juvenile pupfish reach sexual maturity in 3 to 4 months and are generally able to spawn before their first winter (USFWS 2009).

In a study examining Owen's pupfish mating systems and sexual selection, it was found that the size of the mother did not strongly influence egg size or fry size. In addition, individual egg size was not correlated with fry size (Mire and Millett 1994).

Spatial Activity

Little information is known regarding this species' spatial activity. However, CDFW (2013) refers to migration between areas. As noted previously in Habitat Requirements, males are territorial and females occupy areas at the margins of territories.

Ecological Relationships

Generally, the lifespan of Owens pupfish is rarely over 1 year. However, they live up to 3 years in refuge habitats (USFWS 2009).

Owens pupfish congregate in small schools (USFWS 2009). Owens pupfish demography has been studied only in intensively managed refuge habitats with little environmental variation. Demographic studies of other pupfishes in the Death Valley system, however, suggest large seasonal variation in population size. Although studies of Owens pupfish in managed refuge habitats indicate little seasonal variation in population size, unmanaged populations may experience more temporal variation in habitats that are more representative of areas historically occupied (USFWS 2009).

Owens pupfish scarcity in the 1930s was attributed to establishment of non-native predatory fish. In addition, water diversions that decreased and altered Owens River flows desiccated shallow pupfish habitats bordering the river (USFWS 1998).

Population Status and Trends

Global: G1, Critically imperiled (NatureServe 2011, conservation status last updated 2007)

State: S1, Critically Imperiled (CDFW 2013)

By the 1930s Owens pupfish was scarce throughout most of its historical range. It was believed to be extinct from 1942, until in 1964 when a single population of approximately 200 fish was rediscovered in Fish Slough (USFWS 1998). This was the only known existing population when Owens pupfish was listed as federally endangered in 1967. This population still persists today (USFWS 2009). Since its listing, three additional populations have been established at Warm Springs, Well 368, and Mule Springs, (USFWS 2009). These additional existing populations were established from progeny of the remnant population at Fish Slough (USFWS 1998). All existing populations are small, ranging from 100 to 10,000 individuals. The Owens pupfish still faces a high degree of threat, but it also has a high recovery potential (USFWS 2009).

Threats and Environmental Stressors

The 1998 Recovery Plan states that Owens pupfish is affected by non-native species and habitat modification for water diversions that altered Owens River flows (USFWS 1998, 2009). Currently, all populations of Owens pupfish are threatened by loss of habitat resulting from cattail (*Typha* spp.) encroachment. Emergent vegetation and accumulated detritus covers and reduces the substrate used by the pupfish for breeding. Emergent vegetation also reduces water depth, elevates water temperature, and potentially produces severe anoxic conditions (USFWS 2009).

Owens pupfish is also seriously threatened by non-native predators. Because populations are highly localized and relatively small, they can be threatened by a single individual predator. At the time of listing in 1967, several non-native fish predators affecting Owens pupfish were identified: largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), brown trout (*Salmo trutta*), and bluegill (*Lepomis macrochirus*). Since its listing, mosquitofish (*Gambusia affinis*), crayfish (*Pastifasticus leniusculus*), and bullfrogs (*Rana catesbeiana*) have been introduced into the pupfish's habitat and also

threaten Owens pupfish. Besides eating young and adult Owens pupfish, non-native predators compete with Owens pupfish for food and habitat (USFWS 2009).

Additionally, the Owens pupfish is highly vulnerable to extinction from stochastic (random) demographic, genetic, and catastrophic environmental events because the existing populations are small and isolated. Demographic stochasticity refers to random variability in survival and/or reproduction among individuals that can have a significant impact on population viability when populations are small and short-lived with low fecundity (reproductive output). Genetic stochasticity results from the changes in gene frequencies caused by the loss of genetic variation when a new population is established by a very small number of individuals (i.e., the founder effect). This can result in random gene fixation in which some portion of gene loci are fixed at a selectively unfavorable allele (a different form of a gene) because natural selection is not intense enough to overcome random genetic drift. Inbreeding bottlenecks in which a significant percentage of a population is killed or prevented from breeding may also occur in small, isolated populations. Environmental stochasticity is the variation in birth and death rates from one season to the next in response to weather, disease, competition, predation, or other external factors. These three factors may act alone or in combination to reduce the long-term viability of small populations (USFWS 2009).

Conservation and Management Activities

Owen's pupfish reestablishment in the Owens Valley Native Fish Sanctuary has developed as a cooperative undertaking between the City of Los Angeles and the California Department of Fish and Wildlife (CDFW) (Miller and Pister 1971). USFWS and CDFW are making progress toward establishing two new pupfish populations. These populations will be established at the Cartago Springs Wildlife Area (USFWS 2009). Although the four existing Owens pupfish populations do not have approved management plans or implementing agreements between the USFWS and landowners, the new pupfish populations would require management plans that would address threats (USFWS 2009).

Fish screens and the isolation of the artificial refuges for Owens pupfish populations provided some protection from non-native fish

predators. In addition, the CDFW actively removes predators as they are observed. Despite these efforts, predators are likely reintroduced into Owens pupfish populations by fishermen intending to stock those sites with bait and sport fish. Cattail encroachment is currently managed at all populations. If not actively managed, cattails will grow back and threaten Owens pupfish breeding sites (USFWS 2009).

Data Characterization

Few studies have examined the ecology of Owens pupfish. Owens pupfish demography has been studied only in intensively managed refuge habitats that may not be representative of the species' historical, natural demography (USFWS 1998).

Management and Monitoring Considerations

Habitat protection and management is the key to the recovery of the Owens pupfish (BLM 2011). The 5-Year Review for the species includes the following recommendations for actions over the next 5 years (USFWS 2009):

1. Remove emergent vegetation and eradicate non-native predators from Warm Springs and reestablish Owens pupfish in the upper and lower ponds
2. Evaluate Round Valley to determine if it is a suitable location for a population of Owens pupfish
3. Develop management plans and implementation agreements for all populations
4. Establish a new population of Owens pupfish at Cartago Springs Wildlife Area and Blackrock Waterfowl Management Area
5. Conduct population surveys and demographic studies, collect additional genetic samples, and complete genetic analysis. Develop breeding programs based on the results of genetic analysis to optimize genetic material in all populations of Owens pupfish.

Furthermore, the American Fisheries Society has published guidelines for introductions of threatened and endangered species that could be applied to Owens pupfish (Williams et al. 2011). They recommend restricting introductions to sites within the native or historic habitat,

sites that are protected, sites where the potential for dispersal has been determined acceptable, sites that fulfill the species' life history requirements, and sites that contain sufficient habitat to support a viable population. In addition, introduction sites should be avoided where endangered or threatened fish could hybridize with other taxa or where other rare or endemic taxa could be adversely affected. The introduction stock should be from an appropriate source, should be examined for taxonomic status and presence of undesirable pathogens, should be of sufficient number and character, should be carefully and quickly transported, should be introduced under favorable conditions, and the translocation procedures should be documented. After translocation, the American Fisheries Society recommends systematic monitoring of introduced populations, which involves restocking if necessary, determining the cause of any failures, and documenting findings and conclusions reached during the post-introduction (Williams et al. 2011).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens pupfish, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

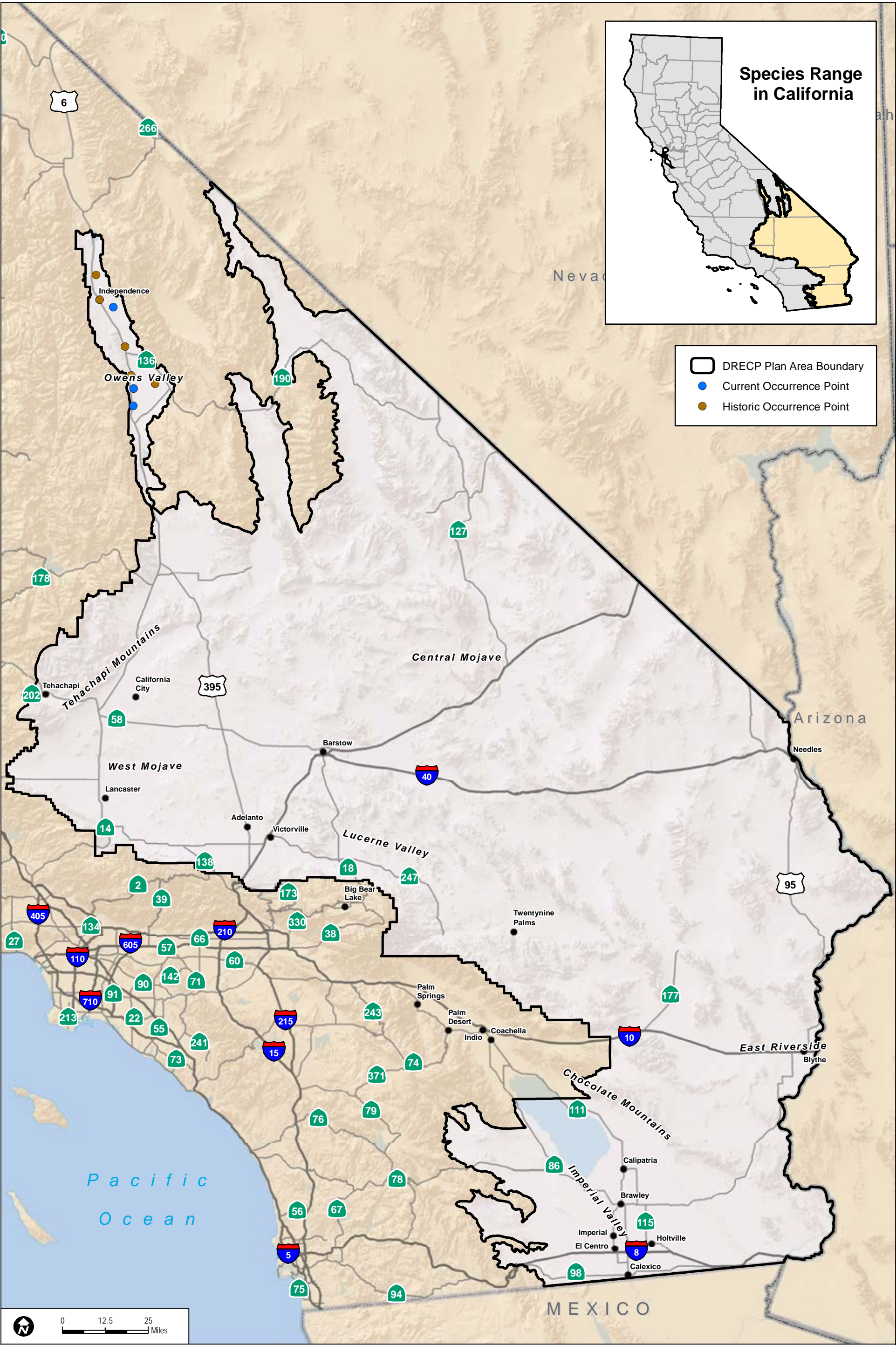
There are 17,547 acres of modeled suitable habitat for Owens pupfish in the Plan Area. Appendix C includes specific model parameters and a figure showing the modeled suitable habitat in the Plan Area.

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-F03
Owen's Pupfish Occurrences in the Plan Area

Owens Tui Chub (*Siphateles bicolor snyderi* = *Gila bicolor snyderi*)

Legal Status

State: Endangered,
Fully Protected

Federal: Endangered

Critical Habitat: Designated on August 5, 1985 (50 FR 31592–31597)

Recovery Planning: Owens Basin Wetland and Aquatic Species
Recovery Plan, Inyo and Mono Counties (USFWS 1998)

Notes: The 5-year review for this species (USFWS 2009) found that threats that were present when the Owens tui chub was listed are still present with new threats identified. The recovery priority number assigned was 3, which indicates the taxon is a subspecies that faces a high degree of threat and has a high potential for recovery (USFWS 2009).



Photo courtesy of Joe Ferreira

Taxonomy

The Owens tui chub (*Siphateles bicolor snyderi*) is a member of the minnow family (*Cyprinidae*). It was described in 1973 as a subspecies of tui chub endemic to the Owens Basin (Miller 1973) as *Gila bicolor snyderi*. Simons and Mayden (1998) published a paper addressing the classification of the North America genera of *Cyprinidae* and, based on ribonucleic acid sequences, restored *Siphateles* from a subgenus to a full genus. The California Department of Fish and Game (CDFG) currently includes the species under the genus *Siphateles* (CDFG 2011), and the U.S. Fish and Wildlife Service (USFWS) proposes the scientific name change from *G. b. snyderi* to *S. b. snyderi* (USFWS 2009). This name change will not affect its federal listing status.

It is morphologically similar to the Mohave tui chub (*S. b. mohavensis*) and Lahontan tui chub (*S. b. obesus*). It is distinguished from its closest relative, the Lahontan tui chub, by scales with a weakly developed or absent basal shield, lateral and apical radii that number 13 to 29, the structure of its pharyngeal arches, the number of anal fin rays, gill-raker

counts of 10 to 14, and 52 to 58 lateral line scales (Miller 1973). Dorsal and lateral coloration varies from bronze to dusky green, grading to silver or white on the belly. The species may reach a total length of 12 inches. The Owens tui chub evolved in the Owens River watershed with only three other smaller species of fishes, Owens pupfish (*Cyprinodon radiosus*), Owens speckled dace (*Rhinichthys osculus* ssp.), and Owens sucker (*Catostomus fumeiventris*) (USFWS 2009).

Based on recent genetic research, Chen et al. (2007) proposed that the Cabin Bar Ranch population is a separate lineage—the Toikona tui chub lineage—from the Owens tui chub lineage. They do not propose making a formal taxonomic split from the Owens tui chub until more information becomes available.

Descriptions of the species' physical characteristics can be found in USFWS (1998) and USFWS (2009).

Distribution

General

The Owens tui chub is endemic to the Owens Basin (Owens Valley, Round Valley, and Long Valley) of Inyo and Mono Counties, California (CDFW 2013; USFWS 1998).

Distribution and Occurrences within the Plan Area

Historical

Early fish collections in the Owens Basin documented Owens tui chub in Owens Lake, several sites along the Owens River from Long Valley to Lone Pine, tributary streams near the Owens River in Long Valley and Owens Valley, Fish Slough, and irrigation ditches and ponds near Bishop, Big Pine, and Lone Pine (Miller 1973; USFWS 2009). Although there are only two historical (i.e., pre-1990) records for Owens tui chub in the Plan Area in the California Natural Diversity Database (CNDDB) (Figure SP-F04) (CDFW 2013; Dudek 2013), the scattered distribution of these localities and the ease with which researchers captured fish suggest that Owens tui chub were common and occupied all valley floor wetlands near the Owens River in Inyo and Mono counties (USFWS 2004).

Recent

Currently, genetically pure Owens tui chub is limited to six isolated sites in the Owens Basin: Hot Creek Headwaters (AB Spring and CD Spring), Little Hot Creek Pond, Upper Owens Gorge, Mule Spring, White Mountain Research Station (operated by the University of California), and Sotcher Lake, the last of which is outside the historical range of the species in Madera County (USFWS 2009). However, there are only three recent occurrence records documented in the CNDDDB database (Figure SP-F04) (CDFW 2013; Dudek 2013). In 1987, Owens tui chub were found occupying irrigation ditches and a spring at Cabin Bar Ranch on the southwest shore of Owens Dry Lake, and became known as the Cabin Bar Ranch population (USFWS 2009). Predation from introduced largemouth bass (*Micropterus salmoides*) and bluegill sunfish (*Lepomis macrochirus*), and failure to maintain adequate water quality and quantity, extirpated the Cabin Bar Ranch population of Owens tui chub in 2003 (USFWS 2009). However, prior to extirpation, 24 individuals were placed in an artificial pond and moved to Mule Spring in 1990; all extant fish of this group descend from this transplant (Chen et al. 2007). The Plan Area only includes the former Cabin Bar Ranch population, with the Mule Spring population (see Figure SP-F04) adjacent and outside of the Plan Area boundary. USFWS (1998) has proposed two conservation areas within the Plan Area: Black Rock and Southern Owens Dry Lake (the Cabin Bar Ranch population was found on the southwest shore of Owens Dry Lake).

Natural History

Habitat Requirements

The Owens tui chub occurs in low-velocity waters with well-developed beds of aquatic plants, rocks, and undercut banks with bottoms of gravel (Leunda et al. 2005; Moyle 2002). Dense aquatic vegetative cover is likely important to Owens tui chubs for predator avoidance, reproduction, water velocity displacement, and feeding (McEwan 1989, as cited in Geologica 2003; McEwan 1991). Plant species observed in occupied habitat at the Hot Creek Headwaters population include watercress (*Nasturtium officinale*), water fern (*Azolla filiculoides*), duckweed (*Lemna* sp.), pondweed (*Potamogeton* sp.), aquatic buttercup (*Ranunculus aquatilis*), and elodea (*Elodea canadensis*) (McEwan 1991).

McEwan (1991) provides details of the habitat structure at the Hot Creek Headwaters population, where plants cover approximately 50% to 75% of the stream surface area. The plants typically grow out from the sides in the main channel, forming dense beds along the stream margins that delineate a small chute of swift-flowing water in the center of the channel. In the backwater areas with zero water velocities, vegetation covers nearly 100% of the surface area. There is a limited die-off of vegetation beds during the winter, but most of the beds persist due to the thermal characteristics of the headsprings.

Water temperature within occupied habitat varies to a great degree (as summarized in Geologica [2003]). It can be fairly constant at spring sites (14–18°C [57–64°F]), hotter at hot springs (21–25 °C [70–77°F]), and cooler in a river (36–78°F [2–25°C]) (Geologica 2003). Within occupied habitat where measurements exist, pH ranges from 6.6 to 8.9 (McEwan 1989; Geologica 2003), dissolved oxygen varies from 5 to 9.3 milligrams/liter (Malengo 1999; Geologica 2003), and alkalinity varies from 68.0 to 88.4 parts per million (McEwan 1989).

The Owens tui chub is restricted to six total populations, five of which are within the historical range of the species. Of these five populations, three (Hot Creek Headwaters, Little Hot Creek Pond, and Upper Owens Gorge) are located in small, isolated, man-altered portions of these waterways. The other two populations (Mule Spring and White Mountain Research Station) exist in manmade ponds at upland sites with water supplied by artificial methods. A detailed account of the habitat at each of the extant populations can be found in the 5-year review (USFWS 2009).

Table 1. Habitat Associations for Owens Tui Chub

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Low-velocity waters	Breeding/foraging	Primary	Low-velocity waters with well-developed beds of aquatic vegetation, rocks, and undercut banks	Direct observation studies

Sources: USFWS 2009; Leunda et al. 2005; McEwan 1991, Geologica 2003.

Foraging Requirements

The results of a gut content analysis indicate that Owens tui chub is an opportunistic omnivore that utilizes a wide variety of food items (McEwan 1991). Aquatic vegetation is especially important as it provides forage and habitat for aquatic invertebrates, the main food item of the Owens tui chub (McEwan 1989, as cited in Geologica 2003; McEwan 1991). Specific food items that appear to be of importance include chironomids, larvae of two species of hydroptillid caddisfly, other aquatic invertebrates, plant material, and detritus (McEwan 1991). There is evidence that the diet varies seasonally at the Hot Creek Headwaters (McEwan 1991); the dominant items in Owens tui chub diet there are chironomid larvae and algae in spring, chironomid larvae in summer, hydroptillid caddisflies in fall, and chironomid larvae in winter (McEwan 1991). Owens tui chubs feed mainly by gleaning and grazing among submerged vegetation (Geologica 2003).

Reproduction

Sexual maturity in Owens tui chub appears dependent on the microhabitat. For example, sexual maturity in springs with constant water temperature has been recorded at 2 years for females and 1 year for males, in comparison to more varied temperatures where males and females reach sexual maturity at 2 years (McEwan 1990, as cited in USFWS 2009). In general, tui chubs congregate from later winter to early summer to spawn over aquatic vegetation or gravel substrates (Kimsey 1954, as cited in Geologica 2003). More specifically, McEwan (1990, as cited in USFWS 2009), recorded spawning from late winter to early summer at spring habitats, and from spring to early summer in riverine and lacustrine or lake-like habitats. Spawning appears to be triggered by day length and warming water temperatures (McEwan 1989, 1990, as cited in USFWS 2009). With the adhesive quality of the eggs, spawning usually occurs over gravel substrate or aquatic vegetation (USFWS 2009). Multiple spawning bouts during the breeding season are likely (Moyle 2002), and females may produce large numbers of eggs at each bout (Geologica 2003). Embryos hatch in 3 to 6 days (Moyle 2002), and may be influenced by water temperature, with eggs hatching earlier in warmer water (Cooper 1978, as cited in USFWS 2009). Larvae remain near aquatic plants after hatching (Moyle 2002). Growth during the

first summer is rapid and slows at maturity, usually in the second to fourth year (Moyle 2002).

Table 2. Key Seasonal Periods for Owens Tui Chub

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			X	X	X	X	X					

Sources: USFWS 1998, 2009.

Spatial Activity

The dispersal, home range, and migratory patterns of Owens tui chub are not well understood. Many of the locations where they are currently found are completely isolated from other populations. Tui chubs congregate from late winter to early summer to spawn over aquatic vegetation or gravel substrates (USFWS 2009). Chen et al. (2007) have determined that the Owens tui chub lineage is more genetically distinct from the Cabin Bar Ranch population (the Toikona tui chub lineage) than the Lahontan tui chub, which may represent independent lines of evolution (i.e., no dispersal). Morphology, swimming ability, and behavior all suggest the species is not adapted to movement through rapid waters (Moyle 2002). Therefore, movement of this species likely requires the presence of vegetation beds so that high-velocity areas are encountered only briefly. Jenkins (1990, as cited in Geologica 2003) observed no Owens tui chub in the Owens River Gorge within riffle habitat. Dispersal of other species of tui chub has been inferred using gene flow, where unidirectional dispersal and bidirectional inter-basin gene flow have been recorded (Chen 2006). In addition, daily migrations have been observed for tui chub in large, deep lakes during summer, whereas they move between deep water during the day and shallow water during the night (Moyle 2002).

Ecological Relationships

Owens tui chub were once common and occupied all valley floor wetlands near the Owens River in Inyo and Mono counties. Since that time, predaceous non-native fishes, extensive development of water resources, and interbreeding with Lahontan tui chub has resulted in population decline and habitat loss.

Currently, the major threat to the species is introgression with Lahontan tui chub (Chen et al. 2007). The Owens tui chub is reliant on slow-moving freshwater habitats that provide food and cover, but that are free of non-native aquatic predators and other tui chub subspecies and hybrids. It requires aquatic vegetation for cover, foraging, and spawning, as well as gravel substrates for spawning. If one or more of these elements are absent, it can be quickly extirpated from a location.

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)

State: Same as above

Within Plan Area: Same as above

Since its listing in 1985, three new populations of Owens tui chub have been established, bringing the current number to six. Four of these populations are in small, manmade or man-altered waters, and one is outside the historical range of the species at an artificial lake (Sotcher Lake). USFWS (2009) recommends that a Recovery Priority Number of 3 be assigned to Owens tui chub, which indicates that the taxon is a subspecies that faces a high degree of threat and has a high potential for recovery. The threats that were present when the Owens tui chub was listed are still present with new threats identified (USFWS 2009).

Threats and Environmental Stressors

USFWS (2009) provides a detailed explanation of the threats to Owens tui chub, which are summarized here. Currently, the major threat to the species is introgression (i.e., hybridization) with Lahontan tui chub (Chen et al. 2007), which has resulted in extirpation throughout most of its range (USFWS 2009). In 1973, the Lahontan tui chub was introduced as baitfish into many of the streams in the Owens Basin. Historically, the Owens tui chub and Lahontan tui chub were isolated from each other, but now hybridization has been documented for populations in Mono County—at Hot Creek (downstream from the hatchery), Mammoth Creek, Twin Lakes–Mammoth, June Lake, and Owens River Upper Gorge Tailbay. In Inyo County, hybridization has been documented at A1 Drain, C2 Ditch, and McNally Canal (Madoz et al. 2005, as cited in USFWS 2009; Chen 2006, as cited in USFWS 2009). If the barriers that are acting to isolate the Owens tui chub populations

from Lahontan tui chub become permeable, this could result in the loss of genetically pure populations of Owens tui chubs at Hot Creek Headwaters, Little Hot Creek Pond, and the Upper Owens Gorge. In addition, the opportunities to establish new populations of Owens tui chub in the Owens Basin are limited by the presence of hybrids in the Owens River and its tributaries. Currently, the only viable locations for establishing the Owens tui chub are isolated springs or the headwaters of streams with downstream barriers to upstream movement of Lahontan tui chubs or hybrids.

USFWS (50 FR 31592–31597) identified extensive habitat destruction and modification as threats to the Owens tui chub, and this is current as of today. Currently, Owens Basin water is in high demand that is expected to increase, which would reduce the overall availability of surface waters. The survival of two populations (White Mountain Research Station and Mule Spring) is dependent upon the continual maintenance of the artificial water supply and assurance of adequate water quality. The Upper Owens Gorge population is a pool created by a beaver dam that is eroding, which is slowly reducing the lacustrine habitat for Owens tui chubs.

Submerged aquatic vegetation is a key habitat requirement for the Owens tui chub, but not with large amounts of emergent vegetation because it may provide cover for nonnative predators of Owens tui chubs, such as bullfrogs and crayfish (*Procambarus* sp.). At the spring sites (Hot Creek Headwaters, Little Hot Creek Pond, and Mule Spring), emergent vegetation (e.g., cattail) have reduced and altered the aquatic habitat, and routine removal of emergent vegetation is required. The Mule Spring and White Mountain Research Station populations require routine management of water quantity and water quality. The environment that the Upper Owens Gorge population inhabits has been severely altered by the construction of a dam, with no mechanism to manage adequate releases of water downstream of the dam.

Since listing, evidence of disease has been observed in some populations of the Owens tui chub (USFWS 2009). In AB Spring at Hot Creek Headwaters, Bogan et al. (2002, as cited in USFWS 2009) found evidence of infection in six of the seven Owens tui chubs that were collected for genetic analysis. Since disease has been identified in Owens tui chubs, it is considered a threat. However, the magnitude of this threat is unknown (USFWS 2009).

The final listing rule (50 FR 31592–31597) identified predation by introduced non-native fish as a major threat to the Owens tui chub. Predation by non-native largemouth bass and brown trout is thought to have eliminated Owens tui chubs from much of their historical range in the Owens River (Chen and May 2003), and it is believed that non-native fish (largemouth bass and bluegill sunfish) played a role in extirpating the Cabin Bar Ranch population (Chen et al. 2007). Mosquito fish (*Gambusia affinis*) may also present a threat, as they are known to prey on small individuals of Mohave tui chub (Archdeacon 2007, as cited in USFWS 2009). At Mule Spring, bullfrogs are present and probably prey on Owens tui chubs, as they are known to prey on other subspecies of tui chubs (Parmenter 2006, as cited in USFWS 2009).

The inadequacy of existing regulatory mechanisms is considered a threat at this time by USFWS (2009), largely due to unregulated actions that could overdraft the aquifer in the Owens Valley Groundwater Basin area, which may result in reduced or no water flow to existing isolated springs and headwater springs of streams in the Owens Basin. The issue stems from the fact that the aquifer in the Owens Basin has not been adjudicated and its use is not regulated. Any reduction in flow from springs in the Owens Basin would result in further reductions of habitat quality and quantity for the Owens tui chub at springs and tributaries of the Owens River.

Currently, Owens tui chub populations are small, between 100 and 10,000 individuals; therefore, random events that may cause high mortality or decreased reproduction could readily eliminate an entire population, which would have a significant effect on the viability of Owens tui chub populations. Furthermore, because the number of populations is small (six) and each is vulnerable to this threat, the risk of extinction is exacerbated (USFWS 2009). The Owens tui chub has experienced population loss from environmental stochastic events and will likely do so in the future. For example, the Cabin Bar Ranch population was lost because of an apparent failure to maintain adequate water quality and quantity and the introduction of non-native predators. Another example is the disappearance of Owens tui chub from the Owens Valley Native Fishes Sanctuary (Fish Slough). Reasons for the loss of this population are not known, but the small, isolated nature of this population likely contributed to their extirpation (USFWS 2009).

In small populations, such as the Owens tui chub, there are a number of factors that may reduce the amount of genetic diversity retained within populations and may increase the chance that deleterious recessive genes are expressed. Loss of diversity could limit the species' ability to adapt to future environmental changes and contributes to inbreeding depression (i.e., loss of reproductive fitness and vigor) (USFWS 2009). Deleterious recessive genes could reduce the viability and reproductive success of individuals. Isolation of the six remaining populations, preventing any natural genetic exchange, will lead to a decrease in genetic diversity.

Conservation and Management Activities

The recovery plan (USFWS 1998) provides a detailed account of management goals that need to be successfully implemented in order for the species to be delisted:

- Establish multiple, self-sustaining populations of Owens tui chubs throughout much of the historical range of the species in six identified conservation areas;
- Ensure these populations are self-sustaining;
- Ensure that each population contains juvenile and three additional age classes, and that the biomass of Owens tui chubs exceed the biomass of deleterious, non-native aquatic predatory species, which would demonstrate successful recruitment and minimal predation on smaller Owens tui chubs by non-native aquatic species;
- Reduce competition with non-native aquatic species;
- Increase the ability to conserve and protect aquatic habitats;
- Implement measures to prevent hybridization with introduced Lahontan tui chubs;
- To the extent possible, reduce the probability of the loss of Owens tui chub populations from stochastic events; and
- Complete an approved management plan and implementing agreement that address water quantity and groundwater management with the land managers.

These recovery plan criteria do not address threats from disease; catastrophic events that may affect the Owens Basin; demographic,

genetic, or environmental stochasticity; or climate change. The recovery plan identifies no recovery criteria for the Toikona lineage, as the occurrence of this lineage was unknown when the recovery plan was approved. The 5-year review (USFWS 2009) finds that none of these management goals has either not been achieved or can't be evaluated.

Data Characterization

The distribution of and threats to Owens tui chub are sufficiently well known to allow coverage of this species in the Desert Renewable Energy Conservation Plan. Missing pieces of information on this species include the lack of understanding of the Toikona lineage as far as origin, genetics, and ecophysiology (Chen et al. 2007). Additionally, the lack of management plans at each of the six existing populations has resulted in less than ideal protections for the species and a poor understanding of the population dynamics. A reintroduction plan with a specific genetic distribution of the current populations is also needed. Considering the degree of known introgression between Lahontan and Owens tui chub (Chen et al. 2007), data on the distribution of genetically pure Owens tui chub and existing barriers is key.

Management and Monitoring Considerations

The Plan Area includes the former Cabin Bar Ranch population at Southern Owens Dry Lake. The Mule Spring population is the closest extant population, which occurs about 2 miles outside the Plan Area boundary. There are also two proposed conservation areas in the Plan Area: Black Rock and Southern Owens Dry Lake. The genetically important and distinct Toikona lineage that occurs at Mule Spring descended from a total of 24 founders from Cabin Bar Ranch and its extant population is confined to two diminutive artificial ponds at Mule Spring (Chen et al. 2007). Chen et al. (2007) have determined that the Owens tui chub lineage is more genetically distinct from the Toikona lineage than the Lahontan tui chub, which illustrates the genetic importance of the Toikona lineage. They have also determined that the Toikona lineage is suffering from low genetic variation that may be a consequence of founder effects. Specific management within the Plan Area may include development of a management plan specific to the Mule Spring population. The management plan should propose

methods to secure the conservation and the management of water quantity, water quality, habitat, and aquatic predators at the existing occupied ponds at Mule Spring. It should also illustrate in detail how to create new populations for the Toikona lineage, as well as increase effective population size. This detail should include a specific standardized genetic protocol. Candidate conservation areas to be evaluated within the Plan Area for new Toikona lineage populations may include Black Rock and Southern Owens Dry Lake. Evaluation criteria may include the presence of suitable habitat and the absence of predators and the Lahontan tui chub and their hybrids. Because so little is known about the Toikona lineage, additional studies and research should be proposed, such as origin, genetics, and ecophysiology.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens tui chub, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 17,384 acres of modeled suitable habitat for Owens tui chub in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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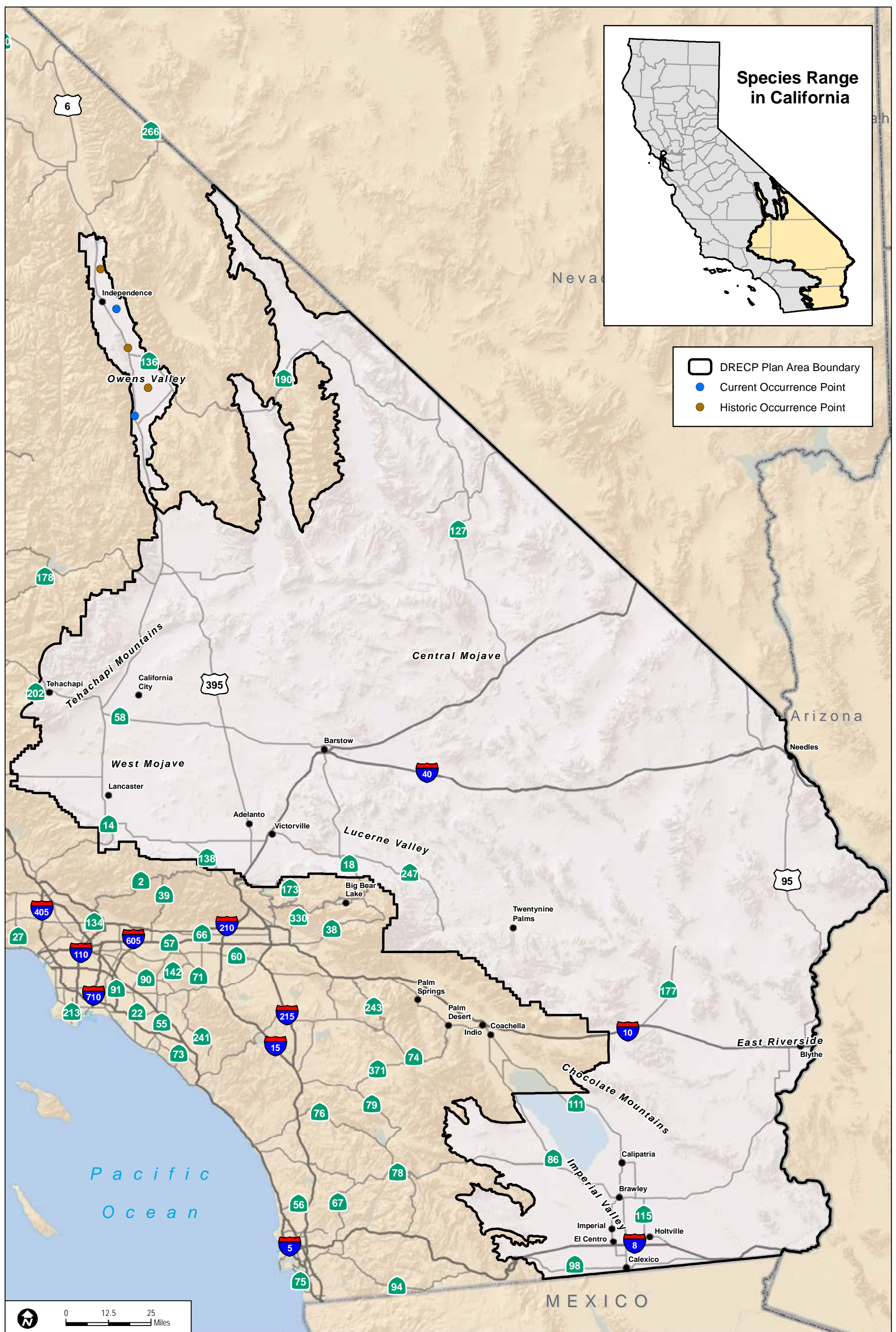
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Burro Deer

(*Odocoileus hemionus eremicus*)

Legal Status

State: None

Federal: None

Critical Habitat: N/A

Recovery Planning: N/A

Taxonomy

The burro deer (*Odocoileus hemionus eremicus*) is the desert dwelling subspecies of the widespread mule deer (*Odocoileus hemionus*). The burro deer was first described by Mearns in 1897 from a specimen taken near the Gulf of California in Sonora, Mexico. Longhurst and Chatting (as cited in Celentano and Garcia 1984) reported that burro deer are distinguished from other subspecies on the basis of cranial measurements, external body measurements, and coloration. Since 1997, desert mule deer (*O. h. crooki*) and burro deer (*O. h. eremicus*) have been synonymized (*O. h. eremicus*) (Heffelfinger 2006). As a result, the overall area identified as containing this subspecies now encompasses much of the southwestern United States and northern Mexico, including southeastern California (Marshall et al. 2004).

Distribution

General

Mule deer are widespread across most of the western United States, western Canada, and south into northern Mexico. The burro deer subspecies is native to the Mojave and Sonoran deserts of the southwestern United States and northern Mexico. Within California, the burro deer is found in the eastern portions of Imperial and Riverside counties, and as far north as the southeastern corner of San Bernardino County. From the Colorado River they range west into California along vegetated washes to the Coxcomb Mountains, Palen Mountains, Little San Bernardino Mountains, Chuckwalla Mountains, Chocolate Mountains, and formerly through the Imperial Valley to

Indio. Burro deer are predominately associated with major river corridors and dry desert washes leading down to the Colorado River and other major rivers. In the hottest months deer are found close to permanent water and forage sources such as the Colorado River. However, with the onset of the summer monsoons in early August and September, burro deer may disperse to the desert mountains (Celentano and Garcia 1984).

Distribution and Occurrences within the Plan Area

Historical

The distribution of burro deer within California was described as far back as 1936 and appears to reflect their current distribution, though it is thought that their former range extended northwest through the Imperial Valley to Indio, and may once have extended around the west side of the Salton Sea (Celentano and Garcia 1984). Much of the area west of Salton Sea and north to Indio was converted to agriculture several decades ago. No pre-1990 occurrences are recorded within the California Natural Diversity Database (CNDDB); however, annual harvest population estimates indicate that the burro deer population fluctuated between 2,000 and 5,000 individuals between 1940 and 1990 (Celentano and Garcia 1984; CDFG 1997, 2007).

Recent

There is no evidence to suggest that burro deer distribution differs from historical (pre-1990) distribution described above. Because burro deer is not a state special-status species, it is not tracked in the CNDDB. However, data compiled by the Conservation Biology Institute (CBI) includes at least six mapped occurrence locations within the Desert Renewable Energy Conservation Plan (DRECP) Area (Figure SP-M02) (Data Basin 2013). Three of the occurrences were along or near the Colorado River, including one near Blythe and the other two in the Palo Verde Area. Two adjacent occurrences are located in the Smoketree Valley area and the other occurrence is near Clemens Well in the valley between the Orocopia and Chocolate mountains. The most recent available estimates made to assist with hunting and herd management put the current burro deer population at about 2,000 individuals (CDFG 2007).

Natural History

Habitat Requirements

The burro deer is a large ungulate that shifts seasonally between desert riparian washes and more open, mountainous terrain. It depends on the availability of water and tracks the best available forage throughout the year. Burro deer need to drink at least every 3–4 days, but tend to drink each night, and therefore require predictable water sources. Consequently, their seasonal distribution is closely associated with water availability (Celentano and Garcia 1984).

During the driest season, between January and March, deer concentrate in lowland riparian habitats, including riparian forest, alluvial and riparian scrub, and alluvial woodland, where water is predictable and forage vegetation quality is relatively high. With the onset of the summer monsoonal rains in July and August, burro deer are less constrained by water sources and use the network of alluvial and wash communities to migrate between lowland riparian communities and the mountainous desert communities that include Sonoran Desert scrub, alluvial woodland, and Joshua tree woodland (Celentano and Garcia 1984; Marshal et al. 2006a) (Table 1). Burro deer remain at high elevations throughout the autumn and winter (Marshal et al. 2006a), only returning to more predictable forage and water sources at lower elevations in spring (Table1).

Burro deer track the highest quality forage, which depends on monsoonal and winter rainfall. Monsoonal rainfall in particular can be highly localized, and consequently forage quality is very heterogeneous (Marshal et al. 2006a, 2006b). As a result, burro deer abundance and distribution can be highly variable from year to year (Marshal et al. 2006c).

Table 1. Habitat Associations for Burro Deer

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Riparian Forest; Alluvial and Riparian Scrub; Alluvial Woodland; Desert Dunes.	Shelter and foraging	Spring, early Summer	Xeroriparian washes, riparian habitats used for shelter and foraging.	Celentano and Garcia 1984; Marshal et al. 2006a
Sonoran Desert Scrub; Alluvial Woodland; Joshua Tree Woodland.	Rutting/fawning/foraging	Summer/Autumn/Winter	Females and fawns steeper slopes, avoiding ridges and valley flats.	Marshal et al. 2006a; Marshal et al. 2006c

Foraging Requirements

Burro deer foraging patterns vary seasonally and are dictated by water availability and quality of forage plants (Marshal et al. 2006a). Their forage is dominated by browse and forbs, with only 10% of their diet consisting of grasses and succulents (Krausman et al. 1997; Marshal et al. 2006b, 2012). During the driest season, in spring and pre-monsoonal summer, burro deer are closely associated with water sources and, consequently, rely on riparian, xeroriparian, and desert wash communities that produce most of the high-quality forage. Forage plants include catclaw (*Acacia greggii*), desert ironwood (*Olneya tesota*), palo verde (*Parkinsonia florida*), honey mesquite (*Prosopis glandulosa*), and cheese bush (*Hymenoclea salsola*). Deer foraging adjacent to the Colorado River include salt cedar (*Tamarix* spp.), cattails (*Typha domingensis*), and arrowweed (*Pluchea sericea*) in their diet (Marshal et al. 2004, 2006b, 2012).

Following the onset of the monsoon between late July and early August, burro deer are less constrained by water sources and are found on steeper ground at high elevations (Marshal et al. 2006a). Common forage plants for burro deer in piedmont and mountainous

areas are creosote bush (*Larrea tridentata*), burro-weed (*Ambrosia dumosa*), brittle-bush (*Encelia farinosa*), and ocotillo (*Fouquieria splendens*) (Marshal et al. 2006b).

As noted above, burro deer forage is dominated by browse vegetation. Microhistological examination of deer pellets found that diets of burro deer had high proportions of browse (76%–85%) in all seasons and low proportions of grasses (1%–2%) and forbs (4%–8%). Browse plants were dominated by saltbush (*Atriplex* spp.), Mexican tea (*Ephedra californica*), desert ironwood, palo verde, and honey mesquite (Marshal et al. 2004, 2012).

Reproduction

Burro deer tend to rut and mate later than most mule deer (Heffelfinger 2006). Rutting and mating may occur as early as late December and as late as March (Table 2) (Celentano and Garcia 1984; Marshal et al. 2006a).

Fawning occurs between July and mid-October (Table 2), timed to take advantage of summer monsoon rains. Fawning occurs in both riparian and mountainous desert habitats, although observations made during fawning indicate that it occurs in areas characterized by low hills with a network of interconnecting washes (Celentano and Garcia 1984). Does with fawns then move into more mountainous terrain where they have a tendency to avoid valley floors and ridges, which are associated with higher predator densities (Marshal et al. 2006a). Fawns are believed to be susceptible to coyote (*Canis latrans*) and golden eagle (*Aquila chrysaetos*) predation until they are at least 6 months old (Marshal et al. 2006a).

Table 2. Key Seasonal Periods for Burro Deer

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Rutting/ Breeding	X	X	X									
Migration							X	X				
Fawning/ rearing of young							X	X	X	X	X	

Sources: Celentano and Garcia 1984; Marshal et al. 2006a

Spatial Activity

Burro deer generally follow a seasonal migratory pattern in the Plan Area. During the drier spring and summer periods, burro deer occur in riparian woodlands and washes bordering major water sources such as the Colorado River, Coachella Canal, or All American Canal. As the summer monsoonal rains arrive, between late July and August, burro deer migrate to the desert mountains, coinciding with the flush of new growth for desert forage plants and raising fawns (Celentano and Garcia 1984). Burro deer only shift back to the lowlands in spring as temporary water sources dry out. Migration is not universal, however, and some burro deer remain around permanent water sources in the Chocolate Mountains (Celentano and Garcia 1984).

Home range patterns vary considerably between seasons. During the hot spring and summer months, deer are restricted to permanent water sources and do not range far. Burro deer occupying Colorado River riparian woodlands may have home range as small as 1 square mile, while deer in dry wash woodland may have home ranges of 2–8 square miles (Celentano and Garcia 1984). During the cooler winter months, when movement is not restricted by water or high temperatures, individual ranges in the mountains may cover 30–50 square miles (Table 3).

Table 3. Movement Distances for Burro Deer

Type	Distance/Area	Location of Study	Citation
Home Range Summer	1–8 square miles		Celentano and Garcia, 1984
Home Range Winter	15–30 square miles		Celentano and Garcia 1984

Ecological Relationships

Rainfall has an important influence on mule deer populations in the deserts of Southern California, with both abundance and population dynamics related to the amount of rainfall. Forage resources in deserts are affected primarily by rainfall, which is highly variable seasonally between years and across the range. As a result, resource availability and its influence on deer populations is highly variable from year to year (Marshall et al. 2002, 2005). Despite these general relationships, however, there is currently no direct evidence linking burro deer population dynamics to the large-scale climatic variation caused by El Niño southern oscillation events (Marshall and Bleich 2011).

During the summer monsoonal season, rainfall events tend to produce strip rains, where a large amount of rain falls on an area about 1 kilometer wide and several kilometers long, with little rain falling on adjacent areas. Strip rains produce a highly heterogeneous response in plant growth (Marshall et al. 2005) and a patchy distribution of forage biomass and quality. Burro deer respond to this heterogeneity by selecting areas with rapidly growing plants, such as those in areas that recently received rainfall, because forage from those plants are high in water, protein, and digestibility. When rapidly growing forage is not available, deer may select areas of high forage biomass, where they can take advantage of forage of higher digestibility before plant biomass and digestibility decrease. When forage water decreases beyond a critical threshold, however, locations of permanent water, including catchments, may become most important in determining deer distribution, and forage growth and biomass become secondary to water availability (Marshall et al. 2005).

It is unclear to what degree mule deer compete or interact with other large- and medium-sized herbivores in the area, such as bighorn sheep (*Ovis canadensis*), feral ass (*Equus asinus*), black-tailed

jackrabbit (*Lepus californicus*), desert cottontail (*Sylvilagus audubonii*), and desert tortoise (*Gopherus agassizii*). Studies assessing the overlap between deer and the feral ass indicate biologically significant overlap, but with the burro deer diet containing more browse and forbs and significantly less grass than the ass (Marshal et al. 2012). Burro deer and bighorn sheep may share diets where their habitats overlap, but they exhibit seasonal separation. In the driest periods of spring and summer, when bighorn sheep may use desert washes, burro deer tend to concentrate in riparian habitats.

Potential predators of burro deer include mountain lion (*Puma concolor*), coyote, bobcat (*Lynx rufus*), and golden eagle. However, the extent to which predators affect burro deer populations is currently unknown. Marshal et al. (2006a) suggest that predators, particularly coyote, may be responsible for females with fawns avoiding valley floors and ridges until the fawns are at least 6 months old. Predator exclusion experiments in Arizona have shown that predation is a significant factor in fawn mortality (Heffelfinger 2006).

Population Status and Trends

Global: Secure (NatureServe 2012)

State: Stable

Within Plan Area: Stable

Burro deer are not currently listed as threatened or special status, but are managed in California for their recreational, educational, and hunting value. Available evidence suggests the population is stable. Past surveys estimated a population of about 2,000 individuals (Celentano and Garcia 1984), with estimates in the 1980s and 1990s varying between 2,000 and 5,000 individuals (CDFG 1997). More recent estimates in the early 2000s from telemetric and remote photographic studies estimate herd densities of 0.05–0.13 deer per square kilometer (Marshal et al. 2006c), indicating a population in the range of 970 and 2,500 individuals.

For hunting purposes, population trends and herd health have generally been inferred from harvest data, climatic conditions, and plant productivity (Celentano and Garcia 1984). However, deer harvests observed a fourfold increase between 1948 and 1998 (Marshal et al. 2002). Such an increase is a reflection of increased hunting intensity and

changes in reporting methods for harvested deer (Celentano and Garcia 1984; CDFG 1997). The increased hunting intensity has, thus far, had no detectable effect upon the population. Current population size and composition are estimated from harvest models, developed in the mid-2000s. The most recent available estimate for 2007 puts the population close to historical levels: 1,940 individuals in 2007 compared to 2,000 individuals in 1940 (CDFG 2007).

Estimates of herd composition are highly variable (Table 4). Celentano and Garcia (1984) estimated sex and age ratio using aerial and ground telemetry, and Thompson and Bleich (1993) tested the efficacy of ground, aerial, and hunter surveys in estimating herd composition but did not estimate abundance. The most recent population estimates for the East Chocolate-Cargo Muchacho area concluded that burro deer occur at densities between 0.05-0.13 deer per square kilometer. This estimate is comparable to the historical estimates of deer densities of 0.08 deer per square kilometer in 1940 and 0.11 deer per square kilometer in 1952 (Marshall et al. 2006c).

The extensive telemetry and remote photography studies conducted between 1999 and 2004 focused on demographic composition, habitat utilization, and potential interactions with other large herbivores such as feral ass. It is evident from these most recent studies that observed abundance and density are highly variable between years, and consequently estimating long-term trends in herd size and health from just a few years of data is difficult (Marshall et al. 2006a, 2006b, 2006c, 2012; Marshall and Bleich 2011).

Table 4. Estimated Herd Composition Ratios from Three Studies of Burro Deer in California

Year	Female	Young	Male	Method
1981 ¹	100	65	No estimate	Aerial and ground telemetry
1982 ¹	100	56	No estimate	Aerial and ground telemetry
1990 ²	100	25	35	Aerial survey
	100	43	29	Ground survey
	100	35	31	Hunter interviews
1999 ³	100	28	9	Remote photography and aerial telemetry
2000 ³	100	17	33	Remote photography and aerial telemetry
2001 ³	100	10	55	Remote photography and aerial telemetry
2002 ³	100	71	38	Remote photography and aerial telemetry
2003 ³	100	43	40	Remote photography and aerial telemetry
2004 ³	100	85	61	Remote photography and aerial telemetry

¹ Celentano and Garcia 1984² Thompson and Bleich 1993³ Marshal et al. 2006c

Threats and Environmental Stressors

Historically burro deer have faced a range of threats from activities associated with an increasing human population in southeastern California. Development and agriculture along the Colorado River has reduced access to the summer riparian habitats, introduced invasive species such as salt cedar, and reduced the availability of native habitats. In addition, increased recreation development and flood control measures have contributed to reduced available summer habitat.

In areas away from the riparian lowlands, increased recreational use of desert washes by off-highway vehicles (OHVs) has resulted in localized disturbances of burro deer, and effectively has reduced connectivity between riparian and mountain habitats. Other localized impacts include mining operations and energy development (Celentano and Garcia 1984).

Historically, poaching, road kill, and drowning in canals have all been identified as significant sources of mortality, although measures taken to reduce road kill and drowning have had some success in reducing these mortality factors (CDFG 1995).

Competition from non-native grazing animals such as feral ass may represent a long-term pressure in shared habitat (Celentano and Garcia 1984; CDFG 1997). The most recent research confirms significant biological overlap in the diet of both species (Marshall et al. 2012).

Other threats found throughout the southwestern desert region include introduction of non-native pasture plants; overstocking and competition from cattle, domestic sheep, and goats; and extensive oil and gas development. However, as yet, these threats appear to be absent from the Southern California range of burro deer (Heffelfinger et al. 2006; Heffelfinger 2006).

Conservation and Management Activities

Several management activities have been implemented specifically to benefit burro deer, or for other species that also benefit the subspecies.

The 1984 *Burro Deer Herd Management Plan* (Celentano and Garcia 1984) was prepared in response to possible stressors and threats from development, agriculture, poaching, and OHVs. The management plan identified actions to maintain habitat health and connectivity as well as actions to mitigate known anthropogenic sources of mortality. The plan included the following key action points:

- a. Maintain access to riparian habitats in summer by controlling recreational uses of riparian habitats, and ensuring agricultural practices are sympathetic to deer requirements.
- b. Maintain contiguous access between summer riparian habitat and winter mountain habitats by ensuring desert wash systems are maintained and not fragmented by development.

- c. Manage access of OHVs to desert wash habitats in core deer population areas.
- d. Reduce road kill incidences along State Highways 78 and 95 by promoting the construction of fencing and underpasses that allow deer to travel between the Colorado River and mountainous habitats.
- e. Ensure that artificial canal construction uses methods that reduce likelihood of deer drowning; e.g., implementation of 2:1 slopes, use of linear curbing.
- f. Reduce illegal hunting.
- g. Document the effectiveness of water source development, i.e., developing catchments that improve availability of free water. This serves two goals: (1) reduces the reliance of deer on open canals as a water source in the driest parts of the year, and thus reduces the risk of drowning; and (2) improves overall access to water for the wider herd.

Desert Wildlife Unlimited Inc. is also involved in providing and maintaining drinkers for desert wildlife, including burro deer. The organization employs 12,000-gallon fiberglass tanks with a step drinker attached, which require relatively little maintenance (Desert Wildlife Unlimited Inc. 2013).

While historically access to permanent water sources has been viewed as the most significant factor limiting desert wildlife, and improvement of water sources has therefore been a primary goal of conservation management (Celentano and Garcia 1984), water sources may only be a limiting factor in the hottest and driest seasons. Throughout much of the year, herd size limitations may be a function of available forage (Marshall et al. 2006b). More recent management recommendations have focused on methods for improving forage availability.

The burro deer should also benefit from habitat conservation and management measures being implemented by the Lower Colorado River Multi-Species Conservation Program (LCR MSCP 2004). Although the burro deer is not a covered species under the LCR MSCP, one of the conservation measures in the LCR MSCP is to provide replacement riparian habitat, which would benefit burro deer, including removal of tamarisk and replacement with suitable native

habitat. An LCR MSCP conservation goal is to create 765 acres of cottonwood-willow and honey mesquite vegetation.

Data Characterization

Burro deer are generally well studied, at least from the perspective of game management. The burro deer herd is managed for harvesting as part of the broader mule deer population in California. Because of its unique desert habitat and management needs, it is managed within its own Deer Management Unit (D12). Annual harvest records are collected from hunters and used in conjunction with fall herd composition data and spring surveys to predict the available bucks for the next hunting season (CDFG 2007, 2010).

Efforts to quantify burro deer population parameters, including population trends and health, have been more difficult because of low densities and low detection probabilities (Thompson and Bleich 1993). Celentano and Garcia (1984) provided estimates of herd density and habitat utilization, but identified a lack of long-term data pertaining to (a) herd age class and sex composition, (b) effects of predators, and (c) effects of illegal kills.

Subsequent studies largely focused on understanding herd composition and age structure (e.g., Thompson and Bleich 1993; Marshal et al. 2005, 2006c), and on quantifying the relationship between rainfall, forage quality, population fluctuations, and management activities (Marshal et al. 2002, 2006a, 2006b, 2012; Marshal and Bleich 2011). However, explicit studies examining the impacts of predators and poaching on this subspecies are absent from the scientific literature. Further, most of the recent studies have been focused in the east Chocolate–Cargo Muchacho areas, providing little information on the status of the herd across the entirety of its range.

Management and Monitoring Considerations

Ongoing management of burro deer herds includes actions to monitor and maintain habitat quality and connectivity as well as activities to reduce known sources of anthropogenic mortality:

- Management of development within riparian and xeroriparian habitats to ensure access between summer and winter ranges

to riparian habitats and clear migration corridors along desert washes (Celentano and Garcia 1984; CDFG 1994, 1995).

- Ongoing monitoring of the effects of illegal hunting (CDFG 1995).
- Assessment and management of feral ass populations to reduce potential competitive effects (CDFG 1997).
- Assessment and development of alternative forage management and enhancement methods to improve quantity and quality of available forage (Marshall et al. 2006a).

Predicted Species Distribution in Plan Area

This section provides the results of habitat modeling for burro deer, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

The model generated 1,150,569 acres of modeled suitable habitat for burro deer within the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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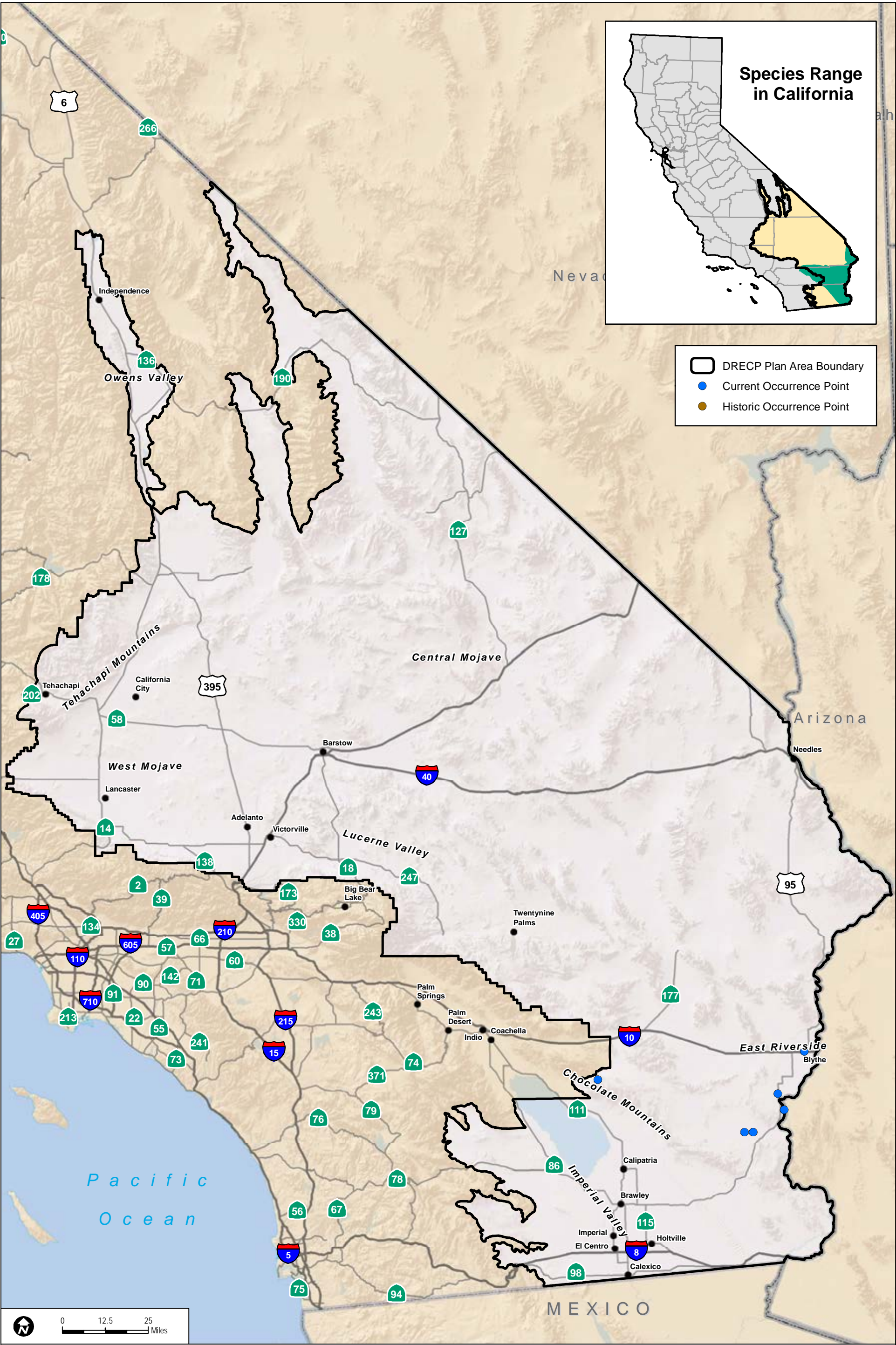
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M02
Burro Deer Occurrences in the Plan Area

California Leaf-Nosed Bat (*Macrotus californicus*)

Legal Status

State: Species of Special Concern

Federal: Bureau of Land
Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Jason Corbett, Bat
Conservation International, www.batcon.org.

Taxonomy

The California leaf-nosed bat (*Macrotus californicus*) is in the family Phyllostomidae and was originally assigned as a distinct full species (Baird 1858, as cited by Rehn 1904). However, based on morphometrics, Anderson and Nelson (1965) placed California leaf-nosed bat as a subspecies of Waterhouse's leaf-nosed bat (*Macrotus waterhousii californicus*), and this was followed by others (e.g., Hall 1981). Based on cranial measurements and chromosomal and biochemical information, California leaf-nosed bat was reassigned to a separate full species *M. californicus* (Davis and Baker 1974; Davis 1973; Greenbaum 1975). Davis and Baker (1974) concluded that *M. californicus* and *M. waterhousii* are "parapatric" species that have contiguous, but non-overlapping distributions. *M. californicus* is currently accepted as a separate species (Wilson and Reeder 2005). A physical description of the species can be found in Wilson and Ruff (Brown 1999).

Distribution

General

The California leaf-nosed bat occurs from southern Nevada and Southern California east to Southern Arizona and south to northern Sinaloa, southwestern Chihuahua, Baja California, and Tamaulipas, Mexico (Wilson and Reeder 2005) (Figure SP-M03). In California, the California leaf-nosed bat occurs in the desert regions of eastern San Bernardino (i.e., excluding the western Mojave region), Riverside, and

San Diego counties and all of Imperial County (Brown and Berry 2004). Although historically the range of California leaf-nosed bats in California reached almost to the southern California coast (Los Angeles/Ventura County line; southern coastal San Diego County, Santa Margarita Ranch [now Camp Pendleton] and DeLuz), the species no longer occurs in these areas, despite repeated searches by bat biologists (Brown and Berry 1998, 2004). Roost disturbance and more important, the loss of suitable foraging habitat have probably led to this regional extirpation (see discussion under Threats and Environmental Stressors). However, even more recent texts do not recognize this loss of range in California in areas outside of the California desert regions that has occurred over the past 60 years (Harvey et al. 2011).

Distribution and Occurrences within the Plan Area

Historical

There are two historical (i.e., pre-1990) occurrences for the California leaf-nosed bat in the Desert Renewable Energy Conservation Plan (DRECP) Area located west of Yuma, Arizona, and north of Interstate 8 (I-8) (Grinnell 1918; Brown et al. 1993a; Brown and Berry 1998, 2004 and 2005; CDFW 2013; Dudek 2013). In writing the bat section of the Bureau of Land Management (BLM) California Desert Plan in 1980, Brown reviewed all historical literature and museum records for bats in the California desert and included her own observations since 1968. (These records occur in the CNDDDB as supplied by BLM regardless of the original source.) Brown and Berry (1998, 2004) surveyed 18 historical sites (records more than 60 years old), and of these, 8 (45%) still sheltered California leaf-nosed bats at the time of the surveys. Howell (1920) also noted that this species was common in caves and mines and that the Salton Sea area supported many caves created by wave action of the sea along its historical coastline. Howell (1920) observed up to 300 individuals in a single colony and collected 63 of them. Arnold (1943) observed the species in the winter in mines and powder magazines near the Laguna and Imperial dams in Imperial County, and Huey (1925) observed a colony of about 500 individuals in a mine shaft north of Potholes in Imperial County. Several historical sites for California leaf-nosed bat occur in San Diego County, including in the Plan Area at the Mollie Mine in Anza Borrego

State Park and a natural cave in Flat Cat Canyon (Banks 1965), as well as the Stage Station at Vallecito and the Artery Mine near Dulzura (Krutzsch 1948) west of the Plan Area. Brown and Berry (1998) visited these areas during the 1980s and 1990s, when assessing the current range for California leaf-nosed bats for the California Department of Fish and Wildlife (CDFW), and no California leaf-nosed bats were found.

Recent

There are numerous recent (i.e., since 1990) records for the Plan Area, including 39 occurrences in the California Natural Diversity Database (CNDDDB) (CDFW 2013) and four roost sites (Figure SP-M03). Brown (pers. comm. 2012) also has provided many records for California leaf-nosed bat in the California desert region. Brown has surveyed more than 2,500 mines or natural caves in 30 mountain ranges in the desert within the range of California leaf-nosed bat over the past 45 years (Brown 1993; Brown and Berry 1998, 2000, 2004). Mountain range extensions (beyond museum and past literature citations) for this species included the Bristol, Marble, Calumet, Eagle, Pinto, Ship, Old Woman, McCoy, Sacramento and Little Maria Mountains in Riverside and San Bernardino counties. Warm mines (and California leaf-nosed bat) have yet to be discovered in other adjacent mountain ranges (Orocopia, Chuckawalla, Little Chuckawalla, Palen, Granite, Coxcomb, Arica, West Riverside, Turtle, Sawtooth, Piute, Clipper, Sheephole and Stepladder Mountains). During a 1995 survey conducted for the Fort Irwin Expansion (Brown and Berry, unpublished data, as cited by Brown, pers. comm. 2012), a few male California leaf-nosed bats were discovered in May in the “Mud Hills” mine at the north edge of the Avawatz Mountains, just south of Death Valley National Park. Guano attributable to this species was also located in a mine near Amargosa Springs. These records suggest a northward extension of the range of California leaf-nosed bat, and the species might occur in the southern part of Death Valley National Park (Brown, pers. comm. 2012).

Natural History

Habitat Requirements

In the California desert, all of the known California leaf-nosed bat roosts are located below 800 meters (2,500 feet) in elevation and most are within 6 kilometers (4 miles) of desert washes containing ironwood (*Olneya tesota*), palo verde (*Parkinsonia* spp.), smoke trees (*Psoralea arguta*) and/or desert willows (*Chilopsis linearis*) (Brown, pers. comm. 2012). The greatest concentration of roosts and those with the largest bat colonies are within the drainage of (and often within sight of) the Lower Colorado River. The roosts discovered near the south end of Death Valley are located in creosote bush scrub. Historical roosts (before development) near coastal areas of California were in chaparral or oak woodland (Brown, pers. comm. 2012).

The California leaf-nosed bat is primarily a cave and mine dwelling species (Anderson 1969; Arita 1993; Arnold 1943; Brown and Berry 2003, 2004; Howell 1920), but also occasionally occupies buildings (Anderson 1969). In Arizona, they have also been found in “open” bridge structures that have cave-like chambers at either end (Davis and Cockrum 1963; Brown and Berry 2004), but most bridge structures are unlikely to be suitable as day roosts. California leaf-nosed bats have been observed using buildings as night roosts east of Searchlight, Nevada (Hatfield 1937) and at Cibola National Wildlife Refuge in California (Brown and Berry 2003). Most winter roost sites in California are mine tunnels at least 100 meters (328 feet) long (Brown 2005). Roost chambers often have large ceilings and considerable fly space (Anderson 1969), although smaller drifts are also used. California leaf-nosed bat is the most northerly representative of the Phyllostomidae, a predominantly Neotropical family. This species neither hibernates nor migrates, and it is incapable of lowering its body temperature to become torpid. Bell et al. (1986) conducted a series of experiments in the laboratory to measure energy metabolism, thermoregulation and water flux to determine if special physiological adaptations allowed California leaf-nosed bats to remain active yearlong in the temperate zone. In the field, daily energy budgets for free-ranging bats were determined using the doubly-labeled water technique. California leaf-nosed bat has a relatively narrow thermal neutral zone, with the lower critical temperature near 34 degrees

Celsius (93 degrees Fahrenheit) and the upper near 37 degrees Celsius (98.6 degrees Fahrenheit). No special physiological adaptations were found in California leaf-nosed bat for desert existence (Lu and Bleier 1981), and they appear to adapt behaviorally rather than physiologically by roosting in geothermally heated winter roosts that have a stable year-round temperature of about 27 degrees Celsius (81 degrees Fahrenheit) (Bell et al. 1986; Brown 2005; Brown and Berry 1998, 2004). Summer roosts may be in more shallow natural rock caves and mines since the summer desert temperatures close to the openings exceed 40 degrees Celsius (104 degrees Fahrenheit) (Brown 2005). Summer roost sites are not always completely dark, and individuals may roost within 10 to 30 meters (33 to 98 feet) of the roost opening. California leaf-nosed bats are tolerant of the highly ammoniated atmosphere of many caves and mines and can tolerate higher concentrations than humans (Mitchell 1963).

California leaf-nosed bats forage in riparian and desert wash areas in California, Arizona, and Nevada (Brown 2005; Huey 1925; Williams et al. 2006) and at tinajas (water-carved natural rock pools) and manmade tanks in southwestern Arizona (Rabe and Rosenstock 2005; Schmidt 1999). Williams et al. (2006) observed California leaf-nosed bats generally using riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland without any apparent differential selection. The tinajas in the Rabe and Rosenstock (2005) study provided open flight approaches and were located near suitable roosting sites (cliffs and rocky canyons). For California, suitable foraging habitats are desert riparian, desert wash, desert scrub, desert succulent scrub, alkali desert scrub, and palm oases (Brown and Berry 2004; Zeiner et al. 1990). In the Sonoran Desert of Arizona (where desert trees are not confined to drainages), a greater percentage of the landscape is utilized by foraging bats (Brown et al. 1999; Dalton et al. 2000; Dalton 2001).

Roosting and foraging habitat associations for the California leaf-nosed bat in the Plan Area are shown in Table 1.

Table 1. Habitat Associations for California Leaf-Nosed Bat

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mines and Caves and occasionally buildings	Roosting	Mines within the California Wildlife Habitation Relationship distribution map boundaries.	Anderson 1969; Zeiner et al. 1990; Brown and Berry 2004
Riparian woodlands desert wash, desert scrub	Foraging	Riparian woodlands, desert wash, desert scrub within 6.2 miles of mines.	Williams et al. 2006; Zeiner et al. 1990; Brown and Berry 2004

Foraging Requirements

California leaf-nosed bat appears to be primarily insectivorous (Anderson 1969). Prey for California leaf-nosed bat include Orthoptera (crickets and grasshoppers), Lepidoptera (butterflies and moths), Coleoptera (beetles), Homoptera (cicadas), and Hymenoptera (ants) (Anderson 1969; Huey 1925; Ross 1961), but at least occasionally takes small vertebrates. Brown (Brown and Berry 2003, 2004) discovered a California leaf-nosed bat in a night roost chewing on the head of a wiggling tree lizard (*Urosaurus ornatus*). Since that time Brown has seen other California leaf-nosed bats carrying tree lizards into night roosts. This reptile spends most of its time in trees and scrubs, often clinging head downward (Stebbins 1985). The California leaf-nosed bat probably gleaned it from the branches of a desert tree when the lizard was sleeping. They are vegetation gleaners and likely take prey directly from the ground or vegetation because some of their prey are flightless and sometimes diurnal (butterflies and lizards) (Stager 1943; Brown and Berry 2004; Anderson 1969; Bell and Fenton 1986). They have short, broad wings that allow them to fly slowly while foraging, with high maneuverability (Anderson 1969; Vaughan 1959), but they are also capable of fast flight with measured speeds of 12 to 14 miles per hour (Dalton 2001; Hayward and Davis 1964). They probably use a combination of echolocation, prey-produced sounds, and binocular vision to locate terrestrial prey (Bell 1985; Bell and Fenton 1986). Their eyes are positioned more anteriorly, and they have superior vision compared to other bats (Bell and Fenton 1986). They usually emerge from day roosts 90 minutes to

2 hours after sunset during the summer and forage in two main bouts during the night (Anderson 1969). During the winter, they may emerge around sunset or shortly after (e.g., within 30 minutes) and forage for about 2 hours (Brown 2005). They may use night roosts that are different from their day roosts (Anderson 1969; also see Hatfield 1937 for use of buildings as night roosts). In the summer, they will roost in desert trees with the foraging area as determined by radio-telemetry (Brown et al. 1999; Dalton et al. 2000).

Reproduction

The largest roosts (over 1,000 individuals of both sexes) are formed in the winter in warm mines. Segregation of males and females usually occurs in the spring and summer, although a few males remain in the maternity colonies. Females congregate in large (>100 bats) maternity colonies, although colonies of only 6 to 20 bats are also found (Barbour and Davis 1969; Vaughan 1959; Brown and Berry 2004). They utilize different mines or areas within a mine separate from those occupied in the winter. Within the larger colonies, clusters of five to 25 females will be associated with a single “harem” male that defends the cluster against intruding males (Brown and Berry 1991). The single young (weighing 25-30% of the mother’s mass) is born between mid-May and early July (following a gestation of almost 9 months) and young are weaned by August (Anderson 1969; Bleier 1975; Bradshaw 1962; Carter and Bleier 1988; Brown and Berry 2004). Since the newborn bats are poikilothermic (a body temperature that fluctuates with the immediate environment), the maternity colony occupies areas close to the mine or cave entrance, where temperatures exceed 32 degrees Celsius (90 degrees Fahrenheit) and daytime summer outside temperatures reach over 49 degrees Celsius (120 degrees Fahrenheit). Most maternity roosts have multiple entrances that allow warm air flow through the mine.

Maternity colonies disband once the young are independent in late summer and breeding occurs in the early fall (Anderson 1969; Brown and Berry 1996). The reproductive cycle of these bats as studied by Krutzsch and others (Krutzsch et al. 1976; Crichton and Krutzsch 1985; Bodley 1974; Bleier 1975; Bradshaw, 1962) shows that viable sperm is not present in the male reproductive tract until August. Ovulation occurs in September and October (Bleier 1971), and unlike many other

bat species that store sperm over the winter and delay fertilization, fertilization occurs immediately after mating, and implantation occurs in later October and November to January (Bleier 1971; Carter and Bleier 1988). Gestation is 8 to 9 months and includes about a 4.5-month diapause period when growth and development is slowed (Bleier 1971; Bleier and Ehteshami 1981; Bradshaw 1962; Crichton and Kruttsch 1985; Crichton et al. 1990). Growth rate and diapause is under control of the hormone progesterone (Crichton and Kruttsch 1985; Crichton et al. 1990). In March, with increased temperatures and insect availability, embryonic development accelerates. Females are reproductively active in their natal year, but males become sexually mature in their second year (Carter and Bleier 1988). Longevity is at least 15 years, based on banding studies (Brown 2005).

In the fall, males aggregate in display roosts and attempt to attract females with a courtship display consisting of wing flapping and vocalizations. The areas used as “lek” sites are usually in or near a mine that had been occupied by a maternity colony (Berry and Brown 1995; Brown and Berry 2004), although exceptions exist. The lek site at Cibola Bridge is located over 11 kilometers (7 miles) from the roost at the Hart Mine (Brown and Berry 2003). In some mines, males defend specific calling areas, while at other sites they will display alongside other males. Aggression between males occurs at this time. Females enter the areas throughout the night, usually roosting in separate groups before approaching a male (Berry and Brown 1995). A banded male observed in the Queen Mine in the Cargo Muchacho Mountains (Imperial County) in September 1994 did not leave the mine during the night, and copulated with at least four females during this period (Brown, pers. comm. 2012). Since the majority of roost surveys have been conducted in the winter and summer, the fall courtship areas for California leaf-nosed bats have not been determined for most mountain ranges.

Key seasonal periods for the California leaf-nosed bat are summarized in Table 2.

Table 2. Key Seasonal Periods for California Leaf-Nosed Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Reproduction				x	x	x	x	x				
Mating									x	x		
Wintering	x	x	x								x	x

Notes: Seasonal migration may occur between mountain ranges.

Sources: Anderson 1969; Bleier 1975; Bradshaw 1962; Brown and Berry 2004

Spatial Activity

California leaf-nosed bats are year-long residents in California (Anderson 1969; Brown and Berry 2004), although historically the species may have migrated to Mexico in the winter (Grinnell 1918) prior to the availability of abandoned mines. Bell et al. (1986) concluded that behavioral adaptations such as foraging methods and roost selection contributed to the successful exploitation of the temperate zone desert by California leaf-nosed bat.

The annual mean temperature in the California desert in the range of California leaf-nosed bat is approximately 23 degrees Celsius (73 degrees Fahrenheit) and the mean winter temperature is 14 degrees Celsius (57 degrees Fahrenheit). All known winter roosts in the deserts of California, Arizona and southern Nevada exhibit stable temperatures greater than 27 degrees Celsius (81 degrees Fahrenheit) and relative humidities above 22%. These mines appear to be located in geothermally-heated rock formations of moderate temperature (Higgins and Martin 1980). California leaf-nosed bats inhabit a stable warm environment (except during their short winter foraging periods). Roost site use does vary seasonally, however, with mixed male/female roosts in the winter and mostly segregated, large, female maternity roosts and smaller, dispersed male roosts during the spring through summer reproductive season (Anderson 1969; Brown 2005), indicating at least local seasonal movements and roost use related to reproduction. Banding studies conducted over the past 43 years suggest that distances traveled between summer and winter roosts are generally no more than a few miles (Brown et al. 1993b; Brown and Berry 1996). Over 25,000 California leaf-nosed bats from

mine roosts along the Colorado River from Parker Dam to Yuma were banded. On yearly trips, usually in the winter, many of these bats were recaptured up to 10 times with an average 50% recapture success rate, suggesting strong roost fidelity, although seasonal movements do occur between roosts. The longest distance between the site of banding and that of recapture was a movement over two mountain ranges for a linear distance of 87 kilometers (54 miles). The greatest time interval so far between initial banding and recapture is 15 years. Assuming that the bat was born in the spring prior to the winter banding, this would indicate a possible longevity of at least 15.5 years. This record for the species is remarkable because long life in bats is usually attributed in some part to their ability to undergo daily and seasonal torpor (Brown, pers. comm. 2012).

There is some information about spatial activity related to foraging. Vaughan (1959) reported that California leaf-nosed bats forage up to 1.3 kilometers (1 mile). Using radiotelemetry, Brown et al. (1993b) observed foraging in desert wash within 10 kilometers (6.2 miles) of roost sites. although more recent data documents captures of California leaf-nosed bats in cottonwood and willow revegetation sites along the Lower Colorado River over 16 kilometers (10 miles) from any potential roosting habitat (Calvert 2009a, 2009b, 2010). As observed by Williams et al. (2006), they generally forage in riparian habitats without any apparent differential selection of riparian type. They also forage at open water sites near potentially suitable roosting habitat (Rabe and Rosenstock 2005). Their ability to fly fast suggests that they could forage fairly far from roost sites. In addition, their selection of limited roosting areas (i.e., primarily temperate caves and mines) suggests that they may be capable of flying quite far to suitable foraging areas that support abundant insect prey, even if most activity is near roost sites (e.g., Williams et al. 2006).

Night roosts are occupied by California leaf-nosed bats between foraging bouts, and may have social significance to the colony. Night roosts are often identified by large amounts of guano and culled inedible insect remains (lepidopteran and orthopteran wings). Bats may return to the same mine used during the day, and roost in different areas. Radio-telemetry studies have shown that individual bats have fidelity to certain night roost sites in shallow mines, rock

shelters, buildings, bridges and trees (Brown et al. 1993b; 1999; Brown and Berry 2003; Dalton et al. 2000).

Ecological Relationships

There is some information about ecological associations for the California leaf-nosed bat, but little data for direct or indirect interspecific interactions. It can be found in association with other bat species at roost sites, including pallid bat (*Antrozous pallidus*), Townsend's big-eared bat (*Corynorhinus townsendii*), and myotis species (*Myotis* spp.) in California (Vaughan 1959; Brown and Berry 2003, 2004). Pallid bats and California leaf-nosed bats have similar ecological attributes as both glean large immobile insects and arthropods, and day and night roost in close proximity in mines. Pallid bats cluster in roosts and often use crevices, while California leaf-nosed bats hang alone from the ceiling (Vaughan 1959).

Desert riparian communities are very spatially limited resources used by a large number of bat species. A likely important factor in bat community diversity and ecological relationships in desert riparian areas is resource partitioning. Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce interspecific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-, within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering. Williams et al. (2006) examined foraging activity by California leaf-nosed bats in riparian habitats in southern Nevada that were also used by 14 other bat species, including both resident and migrant species (see Table 1 in Williams et al. 2006 for the list of species detected). Adequate detection data were collected to analyze habitat use by several of the species. These data show that California leaf-nosed bat, Brazilian free-tailed bat (*Tadarida brasiliensis*), western yellow bat (*Lasiurus xanthinus*), and pallid bat exhibit different habitat selection patterns. While California leaf-nosed bat and Brazilian free-tailed bat were riparian habitat generalists, western yellow bat and pallid bat showed strong preferences for riparian woodland (Williams et al. 2006). Six other bats qualitatively showed more activity in one of the four riparian

types (i.e., riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland), indicating some selection. Overall, riparian woodland, which represented less than 1% of the riparian habitat in the study area, was the preferred habitat type (>50% of all bat activity), with riparian marsh the least used, although it was often used by the spotted bat (*Euderma maculatum*). Williams et al. (2006) suggested that habitat preferences by the different bats may reflect preferred insect prey and abundance, indicating a possible basis for resource partitioning. Given that desert riparian communities are a critical resource for bats, the habitat use information provided by Williams et al. (2006) indicates that managing this diverse habitat type, including hydrology and species composition, is important for maintaining a diverse bat community, including suitable habitat for California leaf-nosed bat.

Population Status and Trends

Global: Apparently secure (NatureServe 2011)

State: Vulnerable to imperiled (CDFG 2011)

Within Plan Area: Same as state

Although historical records from 1894 through 1950 place California leaf-nosed bat in more coastal sections of southern California, these sites are not currently occupied (Grinnell 1918; Howell 1920; Constantine, 1961, 1998; Brown and Berry 1998, 2004), representing a loss of almost 50% when polygons are drawn between historical and current roost areas in California. Urbanization, human disturbance of roosts and destruction of foraging areas are probably the primary factors in their eradication from these areas. With possibly one exception, all California leaf-nosed bat roosts are now located in the desert.

The California leaf-nosed bat is a former U.S. Fish and Wildlife Service (USFWS) Category 2 Candidate for listing under the federal Endangered Species Act and is now a Species of Special Concern for USFWS and the CDFW (Brylski et al. 1998), and a BLM and U.S. Forest Service (Region 5) Sensitive Species. The Western Bat Working Group granted it High Priority for its entire range. www.wbwg.org/speciesinfo/species_matrix/spp_matrix.pdf.

Information collected by Ellison et al. (2003) for California leaf-nosed bat suggested that assessing population trends for this species would be a challenge. Ellison et al. (2003) reviewed information for 143 locations in Arizona, Nevada, and California. Counts at occupied sites ranged from 1 to 2,000 individuals. Trends were analyzed for five colonies, including three winter colonies and two summer colonies, and no positive or negative population trend was apparent. They also noted that the number of individuals at roost sites can fluctuate both between and within seasons, so population sampling would need to account for this apparent natural temporal variation. Ellison et al. (2003) noted, however, that many reports lacked careful and consistent documentation of surveys methods, such as how counts were made, what type the colony was, etc. More recent censuses using standardized methods has revealed stable colony sizes for California leaf-nosed bats in the largest colonies. Over the last 10 to 12 years Brown has conducted censuses by counting exiting bats in the evenings with night vision equipment in the same manner and at the same times of year in the absence of moonlight (Brown 2011). These are usually done in the winter (January or February) when the largest colonies form and for maternity colonies in mid-April or May (prior to young of the year flying). Moon phase was recognized as a significant variable in determining population size by exit counts for California leaf-nosed bat in January 2003 when paired counts were conducted during the week before and after the full moon on selected mines in southeastern California (Brown and Berry 2004; Brown 2011). There was a several-fold increase in the number of bats exiting the mine in the hour after dark in the absence of moonlight. These studies by Brown underscore the need for standardized census methods and consideration of detectability factors to document any population trends.

Threats and Environmental Stressors

The two main threats to this species likely are (1) disturbances of roost sites due to human entrance, abandoned mine closures, and renewed mining in historic districts (Brown 2005; Zeiner et al. 1990) and (2) loss and degradation of desert riparian habitats (Brown 2005). Brown (Brown 2005; Brown and Berry 1998, 2004) cites the loss of desert riparian habitat to development of golf courses and residential housing in the Coachella Valley and the “rip rapping” and channelization of

desert washes as a threat to the species. Ground water pumping and road construction that alters drainage patterns can negatively impact microphyll woodland and desert wash vegetation. Another potential threat is direct or secondary poisoning and loss of prey related to pesticide use for agriculture and golf course operations, and other environmental contaminants associated with mining (Clark 1981; Clark and Hothem 1991).

Several recent studies have documented substantial mortality of bats at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). A general review of the wind facility-related literature failed to reveal evidence for, or discussions of, California leaf-nosed bat fatalities or assessed risks at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009; Cryan and Brown 2007; Kuvlesky et al. 2007). This is likely because of the species' limited range in the southwestern United States and, further, because relatively little systematic post-project bat fatality monitoring data have been collected for large wind energy projects in the southwest (Solick and Erickson 2009). However, California leaf-nosed bats in the Plan Area could be at elevated risk of turbine strikes or from other associated causes (e.g., barotrauma) if a wind facility was located within a few miles of a day roost site (where most foraging activity occurs) and strikes would most likely occur during emergence and return to the day roost. Risk of strikes may also be higher when bats are moving between maternity roosts and winter sites in the fall and spring.

Conservation and Management Activities

California leaf-nosed bat is addressed in the West Mojave Plan (BLM 2005) under Alternative A (the Proposed Action – Habitat Conservation Plan). The BLM would implement several conservation measures for California leaf-nosed bat, including:

- Protection of all roosts containing more than 10 California leaf-nosed bats (Notes: The Plan identified one maternity roost and one maternity/winter roost for the species. Also, the Plan refers to “maternity and hibernation” roosts, but California leaf-nosed bats do not hibernate (Brown, pers. comm. 2012) so reference to these roost types was deleted);

- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public;
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, abandoned buildings, or bridges to determine whether significant roost sites are present; and
- Safe eviction of bats at a non-significant roost (i.e., fewer than 10 individuals) prior to disturbance or removal.

BLM would also conduct monitoring and adaptive management for California leaf-nosed bats. Monitoring actions include:

- Determining bat numbers in all significant roosts (defined by BLM for the West Mojave Plan as more than 10 individuals);
- Conducting periodic surveys of mine openings in Pinto Mountains for bats in areas with high potential for containing significant roost sites;
- Determining and reporting the effectiveness of mitigation measures providing for safe exit of bats;
- Reporting take from approved projects that impact bats under to the CDFG and USFWS; and
- Monitoring population numbers using bat houses if installed (Note: Brown (pers. comm. 2012) indicates that California leaf-nosed bats would not use bat houses, but this is included as conservation measure in the West Mojave Plan).

Adaptive management measures include:

- Gating mines where new significant roosts are found;
- Installing bat houses in locations, where appropriate, if populations decline or are threatened (Note: Brown (pers. comm. 2012) indicates that California leaf-nosed bats would not use bat houses); and
- Desert wash vegetation within 3 miles of known or newly discovered maternity and hibernation roosts of California leaf-nosed bats would be protected. Motorized vehicle use of washes in these locations would be assessed on a case-by-case basis to

determine if vehicles harm the desert wash vegetation. If substantial damage from vehicle use is determined to be present, alternative access routes would be developed and the wash routes would be closed or limited. (Note: California leaf-nosed bat does not hibernate (Brown, pers. comm. 2012), but the West Mojave Plan refers to hibernation roosts).

The California leaf-nosed bat is also addressed in two other BLM plans for the California desert. The *Proposed Northern and Eastern Mojave Desert Management Plan* addresses sensitive bats, including California leaf-nosed bat (BLM 2002a). Under the proposed alternative, this plan includes changing the existing “Moderate Multiple Use Classification” to the “Limited” designation for 7,400 acres of public land in the Silurian Hills region, which is known to support extensive habitat for several sensitive bat species. Route designation would occur on these lands, including seasonal limitations and/or closures to sensitive bat values (e.g. active bat maternity roosts).

The *Proposed Northern & Eastern Colorado Desert Coordinated Management Plan Activities* (BLM 2002b), under all alternatives, would require mitigation measures for projects authorized at or within 1 mile of a significant bat roost site, which may include seasonal restrictions, light abatement, bat exclusion, and gating of alternate sites. If bats are to be excluded from an old mine prior to renewed mining, the exclusion must be performed at a non-critical time by a qualified bat biologist. Mitigation plans for large mines would consider retaining some shafts and adits (horizontal or nearly horizontal opening to a mine) or creating new ones as compensation. Also, under the proposed alternative, Bat gates would be constructed on caves or mine roosts only where there is significant potential for negative effects and closure of any route within 0.25 mile of any significant bat roost would be strongly considered.

In addition, as a BLM sensitive species, California leaf-nosed bat is addressed under other land use actions undertaken by BLM. In accordance with BLM’s “6840 – Special Status Species Management” manual, the objectives for sensitive species policy are:

To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the likelihood of and need for listing of these species under the ESA (BLM 2008).

Under this policy, BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

The California leaf-nosed bat is covered as an “evaluation species” under the Lower Colorado River Multi-Species Conservation Program administered by the Bureau of Reclamation (LCR MSCP 2004). The LCR MSCP defines evaluation species as species that could be listed in future years and that could be added to the covered species list during LCR MSCP implementation, but for which sufficient information was not available for LCR MSCP planning area when the plan was prepared. Conservation measures include: (1) conducting surveys for roost sites within 5 miles of the LCR MSCP planning area in Reaches 3–5; and (2) creating habitat near roost sites, including cottonwood-willow and honey mesquite within 5 miles of roost sites.

California leaf-nosed bat is also addressed in the Military Integrated Resource Management Plan (INRMP) for the Marine Air Ground Task Force Training Command, Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in the INRMP, California leaf-nosed bat is provided protection and management considerations for the military training operations at Twentynine Palms. If it is determined to be at risk from training activities, efforts are made to avoid and minimize impacts. For example, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition;
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant; and
- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

There is substantial information for the distribution of California leaf-nosed bat and its use of mines and caves in the Plan Area. Brown has surveyed more than 2,500 mines or natural caves in 30 mountain

ranges in the desert within the range of California leaf-nosed bat over the past 45 years (Brown 1993; Brown and Berry 1998, 2000, 2004).

Management and Monitoring Considerations

The main management consideration for California leaf-nosed bat is the relationship between human activities near active roost sites, (mine entry by recreation, geologists, etc.), and mine closure for hazard abatement or renewed mining (Brown 2005). Removal of desert wash vegetation near a roost will cause declines (Brown and Berry 1995). Management of riparian communities with regard to hydrology and community structure is also an important management concern (Williams et al. 2006). Pesticide use in agricultural areas or golf courses adjacent to suitable roosting and foraging areas should be managed to prevent potential direct and indirect poisoning and secondary impacts on prey.

Predicted Species Distribution in the Plan Area

This section provides the results of habitat modeling for California leaf-nosed bat, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 8,046,536 acres of modeled suitable habitat for California leaf-nosed bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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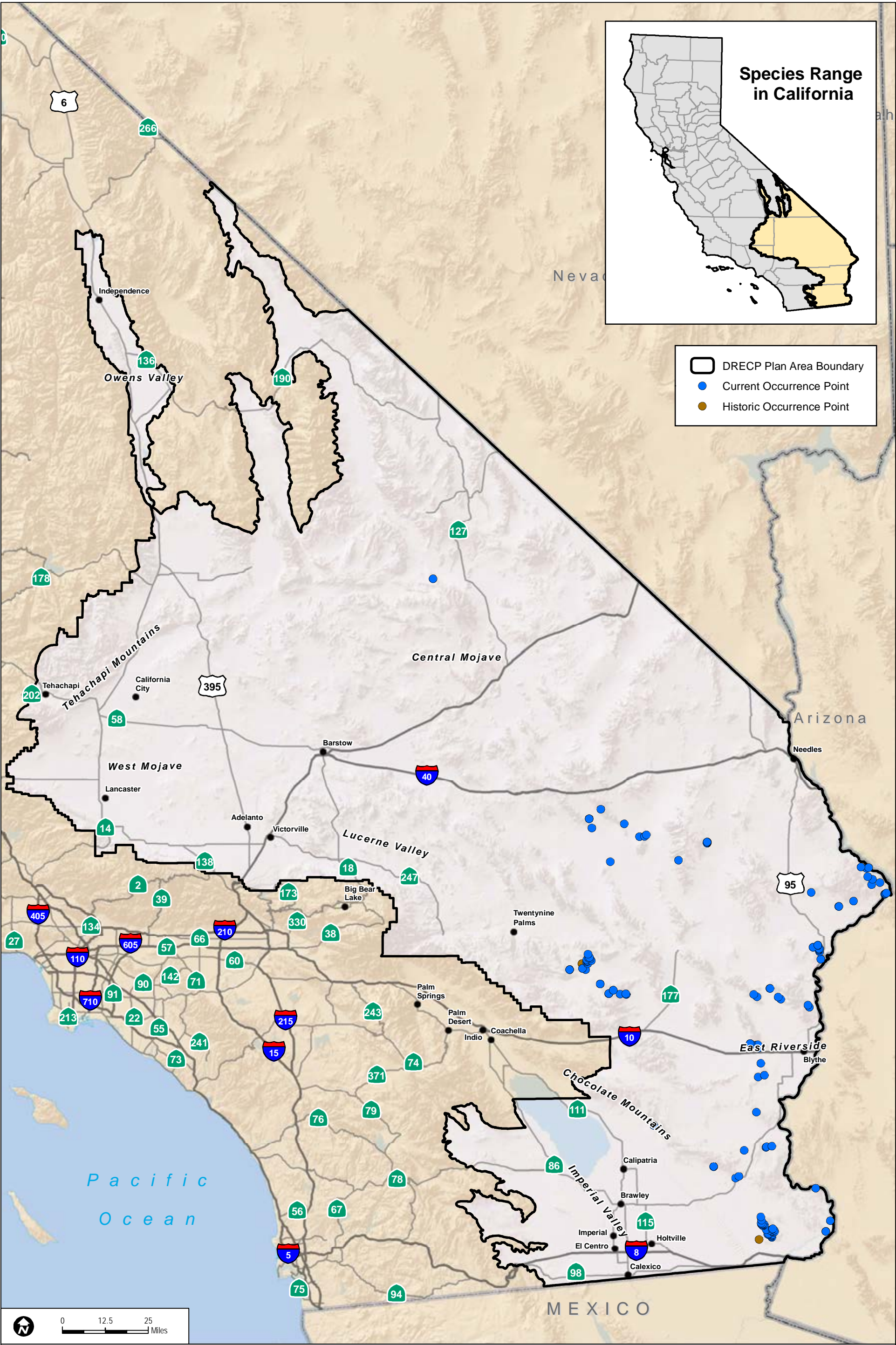
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M03

Leaf-nosed Bat Occurrences in the Plan Area

Desert Renewable Energy Conservation Plan (DRECP) Baseline Biology Report

October 2015

Desert Bighorn Sheep (*Ovis canadensis nelsoni*)

Legal Status



Photo by Dee E. Warenycia.

State: None for subspecies *Ovis canadensis nelsoni* (Nelson's bighorn sheep); Peninsular bighorn sheep distinct population segment (DPS) is Threatened, Fully Protected

Federal: Peninsular bighorn sheep DPS is Endangered; Nelson's bighorn sheep is Bureau of Land Management Sensitive, U.S. Forest Service Sensitive

Critical Habitat: Designated for Peninsular bighorn sheep DPS occupying the Peninsular Ranges of Southern California on April 14, 2009 (74 FR 17288–17365).

Recovery Planning: A Recovery Plan for Peninsular bighorn sheep in the Peninsular Ranges of California was approved October 25, 2000 (USFWS 2000).

Taxonomy

The subspecific taxonomy of bighorn sheep (*Ovis canadensis*) at the subspecies level in the southwest desert region has been uncertain. Earlier studies had placed desert bighorn sheep in one of four subspecies occurring in the southwest desert region (Cowan 1940). For populations within the Desert Renewable Energy Conservation Plan (DRECP) Area, based on cranial measurements, desert bighorn sheep in the Peninsular Ranges were considered a separate subspecies, *O. c. cremnobates*, and northerly populations were designated *O. c. nelsoni* (Nelson's bighorn sheep). More recent genetic and morphometric information does not support the distinct subspecific delineation of *O. c. cremnobates* and the current classification has Nelson's bighorn sheep as the only bighorn subspecies occurring in the Plan Area. Research has found north-south and elevational variation in life history patterns of Nelson's bighorn sheep that tracks differences in temperature regimes in California and on a larger geographic scale (Wehausen 2005, 2006) but with no clear boundaries that might be used to define subspecies.

This clinal variation supports Ramey's (1995) suggestion that all desert bighorn sheep be recognized as one polytypic subspecies. Wehausen (2006) suggested that such regional variation be recognized and considered in conservation planning.

In the 2009 federal critical habitat designation, desert bighorn sheep in the Peninsular Ranges are treated as a DPS of the Nelson's bighorn sheep, and are no longer referred to as a separate subspecies (74 FR 17288–17365). This DPS is federally listed as endangered and state-listed threatened and fully protected. Consistent with the federal critical habitat designation, the common name Peninsular bighorn sheep is retained in this species profile where the information pertains specifically to the federally and state-listed DPS. The common name desert bighorn sheep is used elsewhere where this distinction is not made, but this information for desert bighorn sheep would also apply to the Peninsular bighorn sheep DPS.

Distribution

General

Desert bighorn sheep occur in the desert mountain ranges from the White Mountains in Mono and Inyo counties, south to the San Bernardino Mountains, then southeast to Mexico (Wehausen 2006; Shackleton 1985) (Figure SP-M01). An isolated population occurs in the San Gabriel Mountains (Zeiner et al. 1990). Beyond California, its range extends into southern Nevada, southern Utah, southwestern Arizona, and northwestern Mexico and Baja California, Mexico (Shackleton 1985). Although desert bighorn sheep has a broad overall geographic range, actual populations within the range are scattered and discrete (Shackleton 1985).

The Peninsular bighorn sheep DPS generally occurs in the Peninsular Ranges from the San Jacinto and Santa Rosa ranges south into Mexico. The DPS critical habitat is located in Riverside, San Diego, and Imperial counties (74 FR 17288–17365). The bighorn sheep in this region are restricted to the east-facing, lower elevation slopes below about 1,400 meters (4,593 feet), and most occur at elevations between 91 and 1,219 meters (300 and 4,000 feet) (63 FR 13135).

Distribution and Occurrences within the Plan Area

Historical

All of the California Natural Diversity Database (CNDDB) occurrences of desert bighorn sheep, excluding the Peninsular bighorn sheep DPS, within 5 miles of the Plan Area are historical (i.e., before 1990). These occurrences range from the Last Chance Range near the northeastern portion of the Plan Area south to the Chocolate Mountains in the southeastern portion of the Plan Area. Records marking the eastern boundary of the CNDDB records are from near Straw Peak, the Newberry Mountains, and the San Bernardino Mountains east of Joshua Tree National Monument (CDFW 2013).

Five of the six CNDDB records for Peninsular bighorn sheep within 5 miles of the Plan Area are historical. All of these records lie west of the southern portion of the Plan Area, three are within Anza-Borrego Desert State Park, one is near In-Ko-Pah Gorge, and one is east of San Bernardino National Forest (CDFW 2013).

Recent

The California Department of Fish and Game (CDFG)(2010a) prepared the *Biennial Report to the Legislature Regarding Desert Bighorn Sheep Management* pursuant to Section 4094 of the California Fish and Game Code. This report summarizes census information related to long-term management of desert bighorn sheep (including the authorization of hunting tags) and includes sheep counts in specific management units in 2009 and 2010. The distribution of desert bighorn sheep is grouped by a regional system of subpopulations (or metapopulations) based on natural physical features such as geography and vegetation that affect species occurrence, as well as manmade obstacles that affect distribution, such as freeways (CDFG 2010a). Aerial surveys in 2009 and 2010 documented 1,022 desert bighorn sheep, including ewes, lambs, and rams, in the following mountain ranges: Marble Mountains; Clipper Mountains; Kelso Peak and Old Dad Peak; Clark, Kingston, and Mesquite Mountains; Orocopia Mountains; Sheephole Mountains; South Bristol Mountains; Cady Mountains; White Mountains; and San Gorgonio Mountains. The 1,022 individuals represent minimum populations in these areas because they were the only animals actually observed; population size is

assumed to be larger (CDFG 2010a). The CDFG (2010a) report included the Peninsular bighorn sheep metapopulation, with an estimate of about 950 adults and recruited lambs among the nine distinct subpopulations as of December 2010. Population sizes and trends throughout the species' range in the Plan Area are discussed in more detail in the "Population Status and Trends" subsection.

There are 35 recent occurrences of the Peninsular bighorn sheep DPS in the Plan Area and 13 occurrences just west of the Plan Area (Dudek 2013). These occurrences are clustered in the extreme southwestern portion of the Plan Area (Figure SP-M01).

Natural History

Habitat Requirements

Desert bighorn sheep are mobile and wide-ranging and require a variety of habitat characteristics related to topography, visibility, forage quality and quantity, and water availability (USFWS 2000). Desert bighorn sheep prefer areas on or near mountainous terrain that are visually open, as well as steep and rocky (Wehausen 2006). Steep, rugged terrain is used for escape and lambing. Alluvial fans and washes in flatter terrain are also used for forage and water and as connectivity habitat between more rugged areas. However, based on an assessment of radiotelemetry data, Epps et al. (2007) found that desert bighorn sheep mainly used slopes greater than 10% in intermountain habitats. They used 15% slope as a cutoff value in a model for 'effective geographical distance', or EGD, where cells with slopes less than 15% were considered 10 times more costly to cross than cells with slopes greater than 15%. Because desert bighorn sheep predator avoidance is based on vigilance and visual contact, they tend to avoid dense vegetation (USFWS 2000). Peninsular bighorn sheep in particular avoid higher elevations that support chaparral.

Desert bighorn sheep occur in the following habitats (see Table 1): alpine dwarf-shrub, low sage, sagebrush, bitterbrush, pinyon-juniper, palm oasis, desert riparian, desert succulent shrub, desert scrub, subalpine conifer, perennial grassland, montane chaparral, and montane riparian (Zeiner et al. 1990). A wide range of forage resources and vegetation associations is needed to meet annual and drought-related variations in forage quality and availability (USFWS

2000). Seasonal forage available in alluvial fans and in washes provides a diversity of browse during warmer periods that support lactation and thus is important for reproduction and recruitment of lambs. Foraging behavior is described in more detail herein.

Surface water is an important habitat element for desert bighorn sheep, although individuals can survive without drinking surface water (Wehausen 2006). While desert bighorn sheep may drink water in the cool season, in years of poor forage growth, surface water is most important during the May through October hot season, when most females and associated lambs and yearlings live largely within 2 to 3 miles of water. Males join them at these water sources as the hot season progresses with the onset of the breeding season (Wehausen, pers. comm. 2012). In populations in the eastern Mojave Desert (Old Dad Peak, Kelso Mountains, and Marl Mountains), females occur in areas closer to water and more rugged terrain than males (Bleich et al. 1997). Water sources adjacent to escape terrain are preferred and a lack of water may be a limiting factor in the distribution of desert bighorn sheep populations; there are no known large populations in regions lacking water (Wehausen 2006).

Outside the breeding season, males and females commonly occupy different habitats and usually only come together during the rut period (USFWS 2000). Females prefer particularly steep, safe areas for bearing and initial rearing of lambs (Bleich et al. 1997), especially areas of steep limestone if available (Wehausen 2006). Steep topography is not only important for lambing and rearing, but also helps desert bighorn sheep escape from predators (USFWS 2000). Because desert bighorn sheep primarily rely on their sense of sight to detect predators, open terrain with good visibility is critical for protection from predation (USFWS 2000). Males tend to occupy much less rugged habitat during the lambing season (Wehausen 2006).

Table 1. Habitat Associations for Desert Bighorn Sheep

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Alpine dwarf-shrub, Low sage, Sagebrush, Bitterbrush, Pinyon-juniper, Palm oasis, Desert riparian, Desert succulent shrub, Desert scrub, Subalpine conifer, Perennial grassland, Montane chaparral, Montane riparian,	Primary habitat	Year-round	Desert bighorn sheep prefer areas on or near mountainous terrain that are visually open and steep and rocky and that support surface water. Males tend to occupy much less rugged habitat during the lambing season.	Zeiner et al. 1990; USFWS 2000; Wehausen 2006
Alluvial fans and washes	Foraging	During warmer periods/ lambing		

Foraging Requirements

Bighorn sheep are generalist foragers and feed on a wide variety of plant species (Miller and Gaud 1989; Shackleton 1985). For example, Miller and Gaud (1989) documented 121 plant taxa in fecal samples and through direct observations of desert bighorn sheep in a Sonoran Desert habitat in Western Arizona over an 11-year period. However, the composition of their diet varies with season and location (Bleich et al. 1997; Miller and Gaud 1989; Shackleton 1985; Wehausen 2006; 74 FR 17288–17365). They must be able to access the seasonal abundance of plants at various elevations in various habitat types to maximize resources. Desert bighorn sheep adjust their feeding ranges to exploit areas with more nutritive resources, such as within bajadas, early in the season as high-protein grasses emerge. The relationship between nutritive resources, reproductive success, and optimal timing of birth is complex. Lamb survival is strongly related to spring

body growth, so the earlier they are born the more they can grow before forage quality quickly declines in late spring (Wehausen 2005). However, the earlier the birth, the more likely that ewes will have inadequate food quality during late gestation and early lactation (Wehausen 2005.) The factor that controls this relationship is the body condition of the ewes coming into the reproductive season, with ewes in better condition ovulating earlier in the season because they have the condition to withstand the period with lower nutrient resources (Wehausen 2005).

During the reproductive season, nutritious forage is typically concentrated on alluvial fans and bajadas, and in washes where more productive, wetter soils support more herbaceous forage than steeper, drier, rockier soils. These areas, therefore, are especially important food sources during the heat of summer months and in drought conditions (74 FR 17288–17365). For example, Peninsular bighorn sheep browse year-round on shrubs such as burro bush (*Ambrosia dumosa*), small-leaved hoffmannseggia (*Hoffmannseggia microphylla*), desert lavender (*Hyptis emoryi*), globemallows (*Sphaeralcea* spp.), and jojoba (*Simmondsia chinensis*). Grasses such as six weeks threeawn (*Aristida adscensionis*) and red brome (*Bromus rubens*), as well as cacti (*Opuntia* spp.), are primary food sources in the fall (74 FR 17288–17365). Forbs such as native plantains (*Plantago* spp.) and common ditaxis (*Ditaxis neomexicana*) are primary food sources in the spring (74 FR 17288–17365). The Peninsular bighorn sheep diet is about 57% shrub, 32% forbs, 8% cacti, and 2% grasses (USFWS 2000).

Desert bighorn sheep typically stay close (i.e., within 2 to 3 miles) to reliable sources of water during hot summer months and drink large quantities at each visit (USFWS 2000). Desert bighorn sheep have been known to travel at least 10 miles from perennial water sources and typically visit a water source every 2 to 3 days. Sources of water for desert bighorn sheep include rainwater accumulated in natural collection tanks and potholes in rock, natural springs, and vegetation with high water content, such as cacti (74 FR 17288–17365).

Reproduction

The primary desert bighorn breeding season, or rut period, is between August and October in the Peninsular Range (USFWS 2000) and August

and November in west Mojave Desert (Wehausen 2006). The gestation period is about 6 months (range of 171 to 178 days (Shackleton et al. 1984). Desert bighorn sheep tend to have relatively high conception rates, with a reported rate of 77% to 85% (USFWS 2000). The lambing period depends on location and resources available, but generally desert bighorn sheep have a long lambing season (see Table 2 for key seasonal periods). The reported lambing period for desert bighorn sheep generally occurs between January and June, with most lambs born February to April. In the Mojave Desert, lambing occurs somewhat later than more southerly areas and may begin in December and end in June, with a small percentage of births commonly occurring in summer as well (Wehausen 2006). In a study in the Peninsular Ranges, the lambing season extended from February through August, with 87% of the lambs born from February to April (Rubin et al. 2000). Lambs usually are weaned by 6 months of age.

In the Peninsular Ranges, the reproductive age of ewes ranges from approximately 2 to 16 years of age. As the birthing time approaches, ewes seek isolated sites with shelter and unobstructed views to bear their lambs, secluding themselves from other females (USFWS 2000).

Mortality rates are highest in the first year of life and lamb survival (to 6 months of age) varies by group and year (Shackleton 1985; USFWS 2000) and is related to several factors. Reproductive success in ruminants such as desert bighorn sheep is associated with the mother's body weight, access to resources, quality of home range, and age. As discussed above, lamb survival to summer is strongly related to body growth during the spring (Wehausen 2005). Rubin et al. (2000) found that lamb survival in a Peninsular desert bighorn sheep population was related to the time of year that lambs are born, with the highest survival rate for lambs born in February through April, compared to lambs born later. Lamb mortality may also be caused by disease or disease processes complicated by environmental conditions, including habitat modification (USFWS 2000).

Winter precipitation, which is tied to plant phenology and nutrient availability for desert bighorn sheep, is an important factor in lamb survival (Wehausen 2005). In the eastern Mojave Wehausen (2005) found that rainfall in the months of October and February has the greatest effect on diet quality. Fall rainfall is important for initiating the growth of cold-tolerant species, including annuals, herbaceous

perennials, and perennial grasses, and February is important for both the continued growth of cold-tolerant species, but also the growth of cold-intolerant perennial species. Timing of birthing coincides with peak nutrient availability and the amount of rainfall in the October through April period has a strong effect on lamb survival and recruitment rate (Wehausen 2005). A similar pattern was reported by Wehausen et al. (1987) for a Peninsular Range population in the Santa Rosa Mountains where rainfall in November, January and February was significantly positively correlated with lamb recruitment. Elsewhere in the desert bighorn sheep's range, similar patterns have been observed. Douglas and Leslie (1986) found a positive relationship between fall and winter precipitation and lamb recruitment the following year. Douglas and Leslie (1986) determined that 52% of the variability in lamb survival in desert bighorn population in the River Mountains in Nevada over a 12-year period was accounted for by autumn precipitation during gestation.

While precipitation patterns are strongly associated with lamb survival, lower lamb survival has also been associated periods of increased rainfall, complicating the relationship between rainfall patterns and lamb survival. Wehausen (2005) noted that declining survivorship occurs with rainfall over about 23 centimeters (about 9 inches). It has been hypothesized that increased rainfall may be associated with disease; increased standing water causes an increase in populations of *Culicoides* midges, which are a vector for bluetongue and epizootic hemorrhagic disease viruses (USFWS 2000), but Wehausen (2005) indicates that more research is needed to understand this relationship.

Table 2. Key Seasonal Periods for Desert Bighorn Sheep

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	X	X	X	X	X	X	X	X			X	X
Lambing	X	X	X	X	X	X						X

Source: Wehausen 2006.

Spatial Behavior

Desert bighorn sheep exhibit seasonal differences in habitat use patterns (USFWS 2000), and some populations of females may migrate seasonally between mountain ranges (Jaeger 1994). Seasonal migration by desert bighorn sheep may be more common than previously thought (Wehausen, pers. comm. 2012). They tend to concentrate in areas with water during the hot summer months and expand their ranges away from water sources in the cooler, wetter season (USFWS 2000). They also alter their ranges during rutting and lambing seasons (USFWS 2000). Home range size depends on the availability of required resources, such as water, forage, and lambing habitat, and, thus, varies geographically (USFWS 2000). Forage quantity and quality, season, sex, and age also influence home range sizes. Generally, ram home ranges are larger than those of ewes. In the San Jacinto Mountains, based on a fixed kernel method for estimating home range (95% utilization distribution), the average estimated home range size was approximately 9.8 square miles for rams and 7.8 square miles for ewes (USFWS 2000).

The social structure of desert bighorn sheep is matrilineal (based on female associations). They exhibit gregarious and philopatric (remaining in natal area) behaviors (USFWS 2000). However, rams do not show the same level of philopatry as females and tend to range more widely, often moving among groups of ewes (USFWS 2000). At 2 to 4 years of age, young rams follow older rams away from their natal group during the fall breeding period, often returning after this period. Rams may use the same travel routes year after year (USFWS 2000).

Long-distance inter-mountain range dispersal movements are important for desert bighorn sheep, primarily by rams, but also by ewes (Wehausen 2006). Using radiotelemetry, Bleich et al. (1990) documented substantial intermountain movement between mountains in southeastern California. Epps et al. (2004, p. 103) state that "Three apparent natural recolonizations have been observed in recent years. It is possible that additional extinctions and subsequent recolonizations were undetected between survey years," but they do not provide more detail about these recolonizations. Nonetheless, recent information indicates that intermountain movements and natural recolonizations are not rare occurrences (Bleich et al. 1996; Epps et al. 2010). Epps et

al. (2010) analyzed DNA information and found that both native and translocated desert bighorn sheep have colonized “empty habitats.” Wehausen (pers. comm. 2102) reports that additional natural colonizations have occurred in several ranges, including Deep Springs, Coso, South Soda, South Bristol, Iron, Little Maria, and Cushenbury (San Bernardino Mountains). Further, ewe movements to new groups once thought be rare (e.g., USFWS 2000) are now known to be much more common (Wehausen, pers. comm. 2012). For example, 3 of 10 radio-collared females moved from the Marble Mountains to the South Bristol Mountains in 1992 when that vacant range was colonized (Wehausen, pers. comm. 2012). The available information now indicates that over the past 25 years recolonizations have exceeded the extinctions that occurred in the mid-20th Century during a 30-year drought period and during a period when desert bighorn sheep were being adversely affected by human activities (Wehausen, pers. comm. 2012).

Ecological Relationships

Access to forage and water resources in proximity to rugged escape habitat is critical for desert bighorn sheep (USFWS 2000). Because of the nutritive requirements for supporting reproduction and body growth, the quality of forage during these periods is important (e.g., USFWS 2000, Wehausen 2005). As noted previously, lambing recruitment is generally positively correlated with high winter precipitation. Poor quality forage may adversely affect maternal care if ewes are in poor condition and lamb mortality may be increased through malnutrition, thus adversely affecting recruitment (USFWS 2000). Although lack of water may adversely affect lactation, water sources may also attract natural predators such as mountain lion (*Puma concolor*) that prey on all age classes, and coyote (*Canis latrans*) and bobcat (*Lynx rufus*) that prey on lambs (USFWS 2000). Predation may be an important loss in very small populations, including recent transplants (Zeiner et al. 1990). For this reason, it is important to have rugged escape habitat near water sources.

In addition to being sensitive to natural predators, desert bighorn sheep may be in competition with both native and non-native animals such as mule deer (*Odocoileus hemionus*), livestock, and feral burros for water and food sources (USFWS 2000). Competition with mule deer may occur in the more northern bighorn populations, but may

not be as great in the Peninsular bighorn population (USFWS 2000). Cattle, sheep, and goats may be serious direct and indirect competitors for food and water sources, and may also sources of disease (USFWS 2000). Goats in particular can forage in rugged terrain favored by desert bighorn sheep and tend to overgraze, reducing or eliminating available forage for desert bighorn sheep (USFWS 2000). Cattle and desert bighorn sheep use different habitat types for grazing/browsing (Shackleton 1985), but may compete at water sites. Sheep and goats are an issue for the northern bighorn populations due to risk of disease (Wehausen 2006; Wehausen et al. 2011), but are not currently present in the Peninsular bighorn range (USFWS 2000). Present competition with cattle in the Peninsular ranges is also limited due to general absence of cattle from bighorn habitat (USFWS 2000).

Competition with cattle and feral burros in the Mojave Desert for water and food resources may occur, but a true competition between burros and desert bighorn sheep has not been demonstrated (Wehausen 2006). It is also possible that bighorn use of water sources is affected by the presence of the non-native honeybee (*Apis mellifera*) (USFWS 2000).

Domestic sheep are the major disease source for the northern bighorn populations, and sheep contact has been associated with major bighorn die-offs (Wehausen 2006). Goats also may be a disease source for desert bighorn sheep (USFWS 2000). Diseases contracted from domestic sheep and goats are described subsequently in the Threats and Environmental Stressors Section.

Population Status and Trends

Global: Subspecies *O. c. nelsoni* is apparently secure; Peninsular bighorn DPS is vulnerable (NatureServe 2010)

State: Subspecies *O. c. nelsoni* is vulnerable; Peninsular bighorn DPS is critically imperiled (NatureServe 2010)

Within Plan Area: Same as above for Peninsular bighorn DPS.

The 2009 estimate for the northern populations of Nelson's desert bighorn sheep is a population of approximately 4,800 individuals (CDFG 2010a). This compares with an estimated population of 3,737 individuals in 1972 and 4,500 individuals in 2003 (CDFG 2010a).

Although the broad estimate indicates an increasing or at least stable population, local populations have shown more variability, with some local population declines (CDFG 2010a). The most recent CDFW aerial survey counts for the northern populations of the desert bighorn sheep are shown in Table 3. The large majority of the counts are within the Plan Area, with only the White Mountains Management Unit wholly outside of the Plan Area.

Table 3. Aerial Counts of Desert Bighorn Sheep in Specified Management Units for 2009–2010.

Mountain Range	Survey Date	Ewes	Lambs	Rams	Total
Management Units Within Plan Area					
Marble Mountains	October 2009	88	34	65	187
Clipper Mountains	October 2009	13	4	16	33
Kelso Peak and Old Dad Peak	October 2009	95	15	69	179
Clark, Kingston, and Mesquite Mountains	October 2009	45	6	28	79
Orocopia Mountains	September 2009	39	7	21	67
Sheephole Mountains	May 2009	22	3	17	42
South Bristol Mountains	October 2009	44	13	26	83
South Bristol Mountains	October 2010	33	9	30	72
Cady Mountains	September 2009	92	37	38	167
Cady Mountains	October 2010	102	23	49	174
San Gorgonio Wilderness Area ¹	May 2009	48	15	20	83
Subtotal Within Plan Area²		485	116	315	916
Management Unit Outside Plan Area					
White Mountains	March 2009	59	16	31	106
Grand Totals		544	132	346	1,022

¹ The eastern portion of the San Gorgonio Wilderness Area is within the Plan Area. The counts may include desert bighorn using areas west of the Plan Area.

Table 3. Aerial Counts of Desert Bighorn Sheep in Specified Management Units for 2009–2010.

Mountain Range	Survey Date	Ewes	Lambs	Rams	Total
² Subtotal excludes the 2009 counts for the South Bristol and Cady mountains to avoid double-counting.					
Source: CDFG 2010a.					
Note that counts are minimum population sizes because they are based on individuals actually observed during aerial surveys. Population size is assumed to be larger.					

For the Peninsular bighorn sheep, as of December 2010, there were about 950 adults in nine distinct subpopulations north of the Mexican border, which indicates an upward trend since the mid-1990s (CDFG 2010a). The highest population estimate for the Peninsular bighorn was 1,170 individuals in 1974 (CDFG 2010a). Since that time, population estimates north of the Mexican border for adults have been 570 in 1988, 400 in 1992, between 327 and 524 in 1993, 347 in 1994, 276 in 1996, and 334 in 1998 (USFWS 2000).

Threats and Environmental Stressors

The potential impacts of threats and stressors are closely related to the metapopulation population structure of desert bighorn sheep in the Plan Area. Metapopulations are characterized by groups of partially isolated populations (or subpopulations) that are typically connected by emigration and immigration pathways that allow for exchange of individuals (and genetic material) and for colonizations after local extinctions. Desert bighorn sheep exhibit such a metapopulation structure in the Plan Area in that small local populations are largely restricted to steep, isolated rocky mountain ranges that are scattered across the desert landscape and which are separated by substantial expanses of unsuitable habitat (Bleich et al. 1990; Epps et al. 2010). Based on Epps et al. (2003), there are 13 metapopulations in California, of which approximately 8 occur in the Plan Area. Within each metapopulation in the Plan Area, there are separate population groups ranging from 1 population in the San Gabriel metapopulation to 18 populations in the South Mojave metapopulation (see Table 1 in Epps et al. 2003). In the 2004 population inventory, of the most frequent population size classes in the Plan Area were either 0 or 25-100 (see Table 2 in Epps et al.

2003). As discussed in Spatial Behavior, inter-mountain movements are not rare, but conservation of the species in the Plan Area depends on maintaining intermountain habitat connectivity that allows for dispersal and migrations between populations, and recolonizations of empty habitats (Bleich et al. 1990). This intermountain habitat includes “stepping stones” within movement corridors that are not permanent habitat, but which facilitate movement (Bleich et al. 1990).

Desert bighorn sheep are threatened by loss and fragmentation of important habitats (e.g., lambing and feeding areas, escape terrain, water, travel, and dispersal routes), disease (mostly livestock derived), predation, drought, potential resource competition, and negative interactions with humans (63 FR 13136; USFWS 2000; Wehausen 2006). In addition, some of these threats are interrelated and interactive. For example, habitat fragmentation has resulted in loss of genetic diversity (Epps et al. 2005), which can result in reduced fitness and vigor and make desert bighorn sheep more vulnerable to other threat factors or stressors such as disease, drought, and predation. These kinds of threats or stressors to desert bighorn sheep are magnified in the Peninsular bighorn DPS due to reduced population numbers and consequent higher risk of extinction.

Habitat loss and fragmentation as a result of highways and aboveground canals (e.g., portions of the California aqueduct from the Colorado River to western Riverside County) and high densities of human habitation present obstacles to movement of desert bighorn sheep between mountain ranges that can interfere with the natural metapopulation structure of desert bighorn in the Plan Area. There is essentially no migration across the Interstate highways (Wehausen, pers. comm. 2012). These physical obstacles limit the potential for natural colonization of vacant areas and gene exchange among subpopulations, which are critical to metapopulation viability (CDFG 2010a; Epps et al. 2005; Wehausen 2006). Epps et al. (2005) examined 27 separate bighorn populations in the central and southern Mojave Desert and northern Sonoran Desert had a rapid reduction in genetic diversity (up to 15%) in the 40 years or less of anthropogenic isolation. They concluded that these barriers have eliminated gene flow among populations, and that isolated populations could lose up to 40% of their pre-isolation genetic diversity over the next 60 years.

Historically, disease contracted from domestic sheep has probably been the greatest factor in desert bighorn sheep population declines throughout its range in North America (USFWS 2000; Wehausen 2006). Extensive domestic sheep grazing in northeastern California, northern Nevada, southwestern Idaho, Oregon, and Washington, likely lead to the extirpation of all native populations in these regions. In contrast, where domestic sheep grazing has not been economical, such as Canada and Alaska, little change has occurred in the distribution of native sheep (Wehausen 2006).

Wehausen et al. (2011) provide a comprehensive review of experimental research on the risk of respiratory disease transmission from domestic sheep to bighorn sheep (the so-called “contact hypothesis”), including (1) contact trials between bighorn sheep, domestic sheep and other native and domestic animals; (2) inoculation experiments with no animal contact; (3) studies to isolate and identify specific organism (i.e., bacterial strains and other pathogens) that may be responsible for pneumonia in bighorn sheep; and (4) vaccination experiments. Their review found that the experimental evidence supports the contact hypothesis. Contact between domestic sheep and bighorn sheep, as well as inoculation with certain strains of the bacteria *Mannheimia haemolytica* cultured from the respiratory tracts of domestic sheep, has a high probability of causing fatal pneumonia in the bighorn sheep. At least one study also found that *Pasturella multocoda* cultured from a flock of wild and domestic sheep cause fatal pneumonia in bighorn sheep (Callan et al. 1991). As a test of the domestic sheep-bighorn sheep contact hypothesis, contact trials between bighorn sheep and other native and domestic animals produce low disease and mortality rates, indicating that the high disease and mortality rates of bighorn sheep in contact with domestic sheep are not an artifact of captivity (which was an alternative hypothesis) (Wehausen et al. 2011). The studies of specific organisms responsible for pneumonia in bighorn sheep after contact with domestic sheep failed to clearly identify specific causes (possibly due to the complexity of the disease and/or the sensitivity of culturing methods in identifying the sampled microbial community); nonetheless, the research has clearly demonstrated a negative effect of direct contact between bighorn sheep and domestic sheep despite uncertainty of the nature of the pathogen. Finally, vaccinations failed to reduce the spread of respiratory disease and vaccination is

probably not an effective management tool, both because it apparent lack of effectiveness and the logistical challenges in treating wild populations (Wehausen et al. 2011).

Predation is also a significant factor in desert bighorn sheep mortality, with mountain lion being the major predator. In the Kingston, Clark, and Granite mountains, considerable predation by mountain lion has been documented (Jaeger 1994; Wehausen 1996). In the Granite Mountains, mountain lion predation caused a steep population decline in the desert bighorn sheep population, with the population reduced to 8 ewes for a period of 3 years (Wehausen 1996). In this study all mortalities in the first 3 years of the study were from mountain lion predation (Wehausen 1996). Predation abated after the first 3 years of the study and the population rebounded at 15% annually the next 3 years (Wehausen 1996). Areas of the Mojave Desert where mountain lion predation is a threat to desert bighorn sheep also support populations of native or introduced deer, which is the mountain lion's primary prey (Wehausen 2006). At least four radio-collared male desert bighorn sheep in the eastern Mojave Desert were killed by mountain lions; predation of females was not confirmed and only males tended to use habitats with mountain lions (Bleich et al. 1997). In the Peninsular Ranges, predation is also a frequent cause of mortality. Of 61 documented mortalities of radio-collared sheep from 1992 to 1998 between Highway 74 in the Santa Rosa Mountains and the Mexican border, 42 were attributed to mountain lion (USFWS 2000). Another study of mortality conducted from 1991 to 1996 in the northern Santa Rosa Mountains found that predation accounted for 9 of 32 adult desert bighorn sheep mortalities, of which, 8 were due to mountain lion predation and 1 due to either mountain lion or bobcat predation (USFWS 2000). Coyote and bobcat also prey on desert bighorn sheep, but are more likely to take lambs; a study showed that of nine lamb mortalities recorded in 1998 and 1999, five were attributed to coyote or bobcat predation (USFWS 2000).

Prolonged drought periods can also cause population declines (USFWS 2000; Wehausen 2006). As discussed previously, high-quality forage associated with winter precipitation and water sources are important to support reproduction (e.g., USFWS 2000; Wehausen 2005, 2006). Lamb recruitment is reduced during periods of drought because gestation or lactation is disrupted or maternal care by ewes

in poor condition is reduced, leaving the lambs vulnerable to malnutrition and predation. Drought can increase competition with native and non-native species, such as livestock, for food and water sources (Wehausen 2006). Competition for water sources can also increase congregations around water, thus increasing the risk of disease transmission (USFWS 2000). Epps et al. (2004) examined whether local extinctions of historical desert bighorn sheep populations are correlated with regional climate patterns and found that elevation, precipitation, and availability of dependable springs are strongly related to population persistence. They concluded that climate has already affected local extinction patterns and that desert bighorn sheep are vulnerable to the effects of future climate change, especially if precipitation is reduced in association with climate change. However, while observations of local extinctions are consistent with directional climate change, Epps et al. (2004) also noted that natural climate stochasticity cannot be ruled out as a factor, with population expansions during cooler wetter periods and retreats during periods of increase drought frequency and intensity. It is unknown long-term climate change is the cause of current population trends (Epps et al. 2004).

Within the Peninsular Ranges, negative interactions with humans and pets, and other urban-related factors, are a threat to the Peninsular bighorn sheep (USFWS 2000). In addition to loss and fragmentation of habitat due to urban and rural development, more than 30% of mortalities in one study were directly attributable to human activities, including vehicle collisions, poisoning, and entanglement in fences (USFWS 2000). Humans, pets, off-road vehicles, construction activities, and aircraft also can affect desert bighorn sheep behavior (Leslie and Douglas 1980; USFWS 2000). These factors can affect desert bighorn sheep to the extent that essential activities, such as foraging or the use of important areas (e.g., water sources, mineral licks, lambing areas, traditional movement routes), are disrupted, which can affect the viability of populations through reduced lamb recruitment (USFWS 2000). Human activities may also induce physiological stress such as increased heart rate, which can affect the health of desert bighorn sheep individuals and lamb recruitment (USFWS 2000). Impacts related to human activities may also occur in the northern populations. However, with the exception of livestock grazing and some recreational activities, impacts would be expected

to be less frequent or severe due to reduced human activity in the more remote areas occupied by desert bighorn sheep.

Non-native plants used for landscaping, such as oleander (*Nerium oleander*) and laurel cherry (*Prunus laurocerasus*), have been implicated in the poisoning of desert bighorn sheep (USFWS 2000). Tamarisk (*Tamarix* spp.) is highly consumptive of water, reducing critical surface water sources for desert bighorn sheep (USFWS 2000).

Mortality in a desert bighorn sheep population in the vicinity of Old Dad Peak was linked to type C botulinum (*Clostridium botulinum*) poisoning near two artificial water catchments (guzzlers) (Swift et al. 2000). The investigators reconstructed the probable cause of the poisoning as 13 lambs that fell into and drowned in one guzzler tank while attempting to drink from the top of the tank. A hatch cover had become dislodged when the drinker trough was dry because the tank valve was closed. The decaying lamb carcasses served as the substrate for the growth of *Clostridium botulinum*, which other individuals ingested after a rain increased water levels and allowed sheep to drink from the source (Swift et al. 2000).

Conservation and Management Activities

The Bureau of Land Management (BLM), CDFG, state parks, National Park Service, and private non-profit organizations (the Bighorn Institute, the Anza-Borrego Foundation, Society for the Conservation of Bighorn Sheep, and Desert Wildlife Unlimited, Inc.) have planned implemented and/or participated in numerous conservation and management actions that benefit the desert bighorn sheep.

Conservation and management activities undertaken by the BLM to benefit the Peninsular desert bighorn sheep include the following actions identified in the Recovery Plan (USFWS 2000):

- Installation of gap fencing to eliminate cattle grazing from steep terrain and from water sources in canyons
- Reduction in grazing pressure on allotments
- Closure of most routes of travel east of McCain Valley Road, except to private inholdings, to ranchers, and to Carrizo and Sacatone overlooks

- Designation of wilderness study areas and subsequent management for non-impairment of wilderness values
- Designation of Jacumba, Carrizo Gorge, Coyote Mountains, Sawtooth Mountains, Fish Creek Mountains, and Santa Rosa wilderness areas by Congress, with attendant elimination of vehicular access
- Tamarisk control efforts around water sources
- Establishment of the Santa Rosa Mountains National Scenic Area Visitors Center to provide public education
- Financial assistance to the Bighorn Institute during its formative years, as well as land transfer and lease under the Recreation and Public Purposes Act
- Temporary closure to dogs on most lands in the Santa Rosa Mountains National Scenic Area
- Closure of roads into Dead Indian Canyon and Carrizo Canyon
- Designation of Santa Rosa and San Jacinto Mountains National Monument, which will prohibit mining and off-road vehicle use on federal lands, support coordinated land management by federal agencies, and increase the area's funding priority.

The BLM also issued an Instruction Memorandum in 1992 regarding domestic sheep grazing, such that domestic sheep should not be allowed within 9 miles of desert bighorn habitat, except where topographic features or other barriers prevent physical contact. Also, domestic sheep trailed and grazed outside the 9-mile zone in the vicinity of desert bighorn sheep habitat should be closely managed and carefully herded (Wehausen 2006).

CDFG manages desert bighorn sheep populations throughout much of the state through the Desert Bighorn Sheep Conservation Program (CDFG 2010a). In accordance with Section 1801 of the California Fish and Game Code, the state policy is to preserve, restore, utilize, and manage the desert bighorn sheep population. Limited harvest of desert bighorn sheep (excluding the Peninsular DPS and the Sierra Nevada bighorn sheep (*O. c. sierra*) which are fully protected) in selected areas is provided by state law for biologically sound management (CDFG 2010a). Management of desert bighorn sheep includes sport hunting of rams, with a limit on hunting tags for no more than 15% of the ram population

in a single year (CDFG 2010a). As part of the management program, CDFG is required to report the status of management units; summarize counts of individuals in specified management units (see Table 3); report the number of hunting tags issued; summarize unlawful take of desert bighorn sheep; report the number of individuals translocated; and track the environmental impacts of hunting (CDFG 2010a).

CDFG conducts periodic inventories of the distribution of desert bighorn sheep in California in specific management units to assess population trends and provide the basis for issuance of hunting tags (see Table 3 for the 2009–2010 counts).

CDFG has also prepared management plans for a number of the major herds in California. The CDFG Desert Bighorn Sheep Management Program is currently preparing a range-wide management program that will provide a strategy to conserve populations throughout the state (CDFG 2010a). In 2010, draft regional management plans were prepared and submitted for approval for the Cady Mountains and South Bristol Mountains management units (CDFG 2010a). These plans address the following issues (CDFG 2010b, 2010c):

1. The numbers, age, sex ratios, and distribution of desert bighorn sheep within the management unit
2. Range conditions and a report on the competition that may exist as a result of human, livestock, wild burro, or any other mammal encroachment
3. The need to relocate or reestablish bighorn populations
4. The prevalence of disease or parasites within the population
5. Recommendations for achieving the policy objective of Section 4900, which addresses the potential for limited hunting opportunities for desert bighorn sheep.

A management objective of the state conservation program is to re-establish desert bighorn sheep on historical ranges (CDFG 2010a). Since 1983, CDFG has translocated almost 500 individuals (including the Sierra Nevada subspecies *O. c. sierrae*).

CDFG also conducts capture-sample-radio collar-release studies for research purposes. In 2010, 10 individuals were captured-collared-

released in the Santa Rosa and Vallecito mountains, including 9 ewes and 1 ram (CDFG 2010a).

Anza-Borrego Desert State Park supports a majority of the range-wide Peninsular bighorn sheep population in California. Anza-Borrego Desert State Park has been actively involved in the conservation of Peninsular bighorn sheep for 30 years. Specific activities relevant to the DRECP that were identified in the Recovery Plan (USFWS 2000) are as follows:

- Construction of guzzlers to supplement water supplies
- Annual monitoring (conducted for 40 consecutive years; California Department of Parks and Recreation 2009)
- Research into bighorn sheep ecology and threats
- Tamarisk removal from riparian areas within bighorn sheep habitat to enhance water availability and native plant community regeneration (approximately 120 miles of canyons and stream courses had been treated by 2000)
- Seasonal access closure of bighorn sheep watering areas from June 1 to October 1
- Remove feral cattle from bighorn sheep habitat
- Construct gap fencing to keep stray cattle from entering bighorn sheep habitat
- Public outreach, including production of a 15-minute movie “The Bighorn of Anza-Borrego”
- Closure of some areas to vehicular traffic.

The National Park Service has conducted burro removal from their lands in the Mojave Desert, with the goal of removing all approximately 1,300 burros from the Mojave National Preserve between 1998 and 2001. (<http://www.nature.nps.gov/yearinreview/yir98/chapter06/chapter06pg2.html>). Although true competition between desert bighorn sheep and burros has not been demonstrated (Wehausen 2006), burros have caused adverse impacts on native plant communities, wildlife, soils, water quality (<http://www.nature.nps.gov/yearinreview/yir98/chapter06/chapter06pg2.html>).

The Bighorn Institute is a nonprofit organization formed in 1982 that investigates the causes of desert bighorn sheep declines, particularly among Peninsular bighorn sheep. The institute began monitoring radio-collared desert bighorn sheep in the northern Santa Rosa Mountains in 1982 and the San Jacinto Mountains in 1992. Research activities conducted by the institute include the ecology of bighorn populations in the Santa Rosa and San Jacinto mountains, lamb ecology, captive breeding and wild population augments, annual population surveys, and disease research (Bighorn Institute 2011).

The Anza-Borrego Foundation is the nonprofit cooperating association for the Anza-Borrego Desert State Park and is a sponsor for the annual desert bighorn sheep count, which has been conducted from 1971 through 2010.

The Society for Conservation of the Bighorn Sheep (SCBS) is a nonprofit organization established in 1964 that has several programs for restoring desert bighorn sheep (<http://sheepsociety.com/>) in coordination with CDFG and BLM. The SCBS provides labor to help conduct censuses and to establish “drinker” sites and also conducts water monitoring (including remote water monitoring stations that record available water at drinkers and precipitation) and water hauling to supplement water at some sites. SCBS maintains remote trail cameras to monitor wildlife use of water sites. SCBS also has “Area Captains” that volunteer under the auspice of CDFG and conduct inspections of the drinkers twice a year and “Hot Shot Crews” that conduct repair and maintenance at drinkers.

Desert Wildlife Unlimited, Inc. is also involved in providing and maintaining Drinkers for desert wildlife, including desert bighorn sheep (<http://www.desertwildlifeunlimited.com/home/>). They employ 12,000 gallon fiberglass tanks with a step drinker attached, which require relatively little maintenance.

Data Characterization

Data availability for desert bighorn sheep is excellent and represents one of the best population datasets for any managed species in California. In particular, the Peninsular bighorn sheep DPS has been monitored annually since 1971. Furthermore, extensive research on

the ecology of the desert bighorn sheep has yielded an excellent understanding of its habitat and ecological relationships.

The CDFG, State Parks, Anza-Borrego Foundation, and the Bighorn Institute conduct periodic assessments of the desert bighorn sheep populations in California, including portions of the Peninsular bighorn DPS. CDFG assessments are based on historical and current data from ground, waterhole, and aerial surveys that are suitable for estimating population size classes (CDFG 2010a). The Bighorn Institute conducts annual assessments of bighorn populations in the Northern Santa Rosa and San Jacinto mountains, and includes radiotelemetry data to study habitat use, reproduction, survival, mortality, and general ecology (Bighorn Institute 2011). The annual desert bighorn sheep count in Anza-Borrego Desert State Park has been conducted annually since 1971 and includes mid-summer counts of ewes, lambs, male and female yearlings, and rams in about 21 different locations in the park (California Department of Parks and Recreation 2009).

Management and Monitoring Considerations

The CDFG (2010b, 2010c) identified several management and monitoring considerations for desert bighorn sheep, including demography (numbers, age, sex ratios, and distribution of desert bighorn sheep within management units); range conditions; relocation or reestablishment of populations; and the prevalence of disease or parasites.

The BLM West Mojave Plan determined that the best way to ensure the long-term viability of desert bighorn sheep metapopulations would be by preventing further population losses and fragmentation and restoring populations in vacant historical habitat. Natural and induced colonization may require artificial enhancement of populations, such as water developments (Wehausen 2006). Contact between domestic sheep and desert bighorn sheep should be prevented by eliminating or carefully managing sheep grazing in the vicinity of desert bighorn sheep habitat (Wehausen 2006). To ensure reliable water supply during the summer months, key water sources within current and historical desert bighorn sheep habitat should be closely monitored and potentially enhanced. Water enhancement may promote development of large desert bighorn sheep

populations that may produce natural colonists to reestablish populations in vacant habitat (Wehausen 2006). However, because water sources may also enhance the populations of desert bighorn sheep predators, such as mountain lion, coyote, and bobcat, water enhancement should be limited.

The federal *Recovery Plan for Bighorn Sheep in the Peninsular Ranges, California* (USFWS 2000) identified improving adult survivorship as likely the strongest positive influence on Peninsular bighorn population dynamics in the short term. Over the long term, conservation and effective management of conserved lands are needed to recover the Peninsular bighorn sheep. Minimizing adverse effects of human disturbance by preventing further fragmentation is critical to the persistence of ewe groups bordering the Coachella Valley. Maintaining adequate buffers between urban development and Peninsular bighorn sheep habitat, and effective management of human activities within ewe group home ranges is needed (USFWS 2000).

Habitat fragmentation and population isolation has led to decreased genetic diversity in small isolated populations (Epps et al. 2005). Fragmentation of metapopulations from fenced highways, aqueducts, and losses of some populations should not be permitted. Epps et al. (2005) recommend that existing barriers to movement should be mitigated and new highways in desert bighorn sheep habitat should be designed to minimize disruption of connectivity. Fencing near existing drainage undercrossings should be modified to allow access to the undercrossings and construction of overpasses should be considered to reestablish connectivity (Epps et al. 2005).

When reintroduction stock is available, historical habitat should be restocked to maximize connectivity and the number of populations in remaining metapopulations. Although evidence suggests that existing metapopulations can remain viable if adequately managed and intermountain travel corridors are maintained, opportunities to reestablish connections across recent artificial barriers that now define metapopulations should be considered (Wehausen 2006).

Species Modeled Habitat Distribution

The habitat model used for the Plan Area was provided by BLM and depicts mountain ranges and intermountain habitat for desert

bighorn sheep suitable for both supporting local populations (i.e., mountain habitat) and movement (i.e., intermountain habitat). There are 12,872,136 acres of modeled suitable habitat for desert bighorn sheep in the Plan Area, including 7,976,800 acres of mountain habitat and 4,893,423 acres of intermountain habitat.

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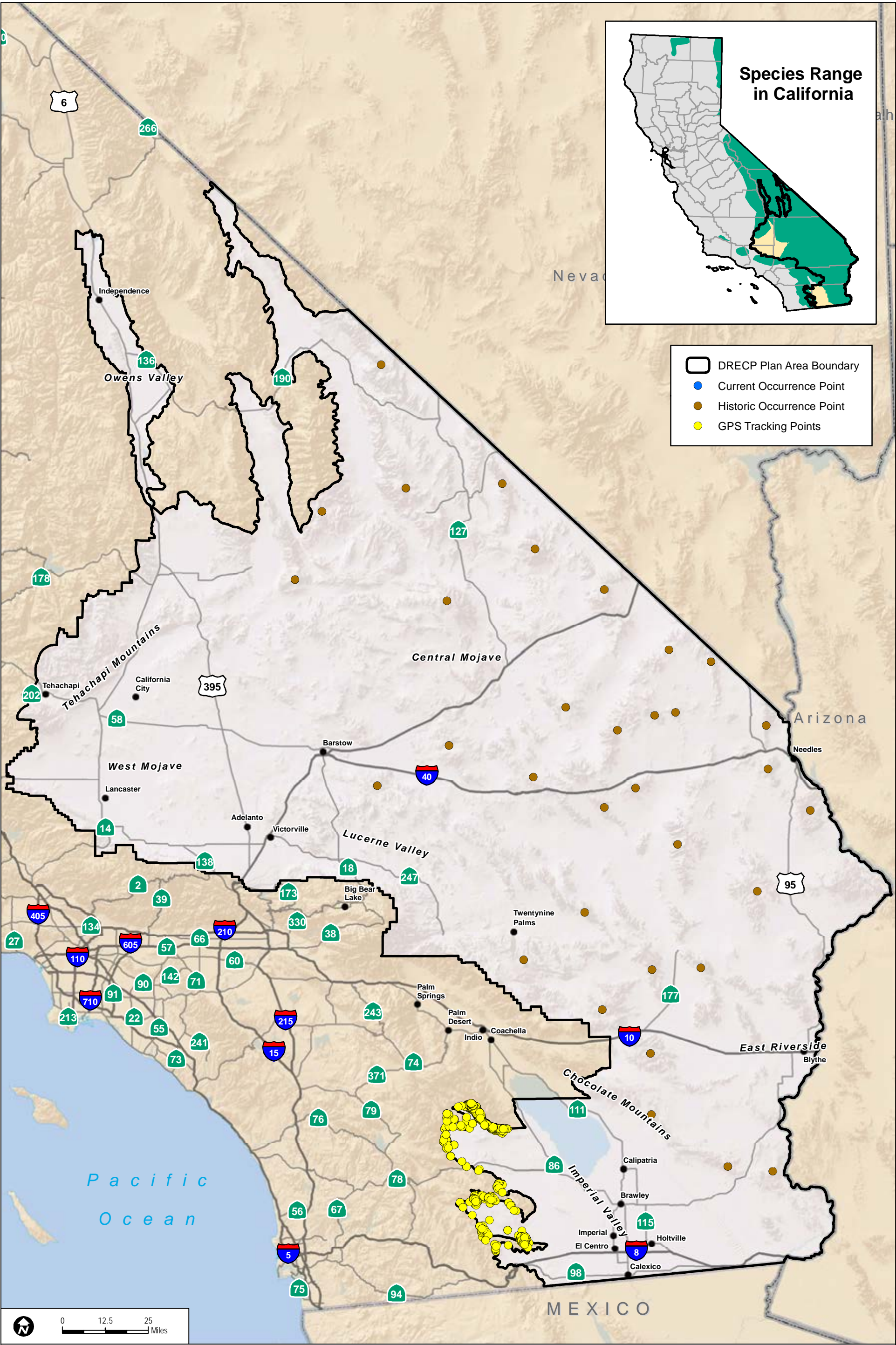
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CDFW (2013); CWHR (2008)

FIGURE SP-M01
Bighorn Sheep Species Occurrences in the Plan Area

Desert Kit Fox

(*Vulpes macrotis arsipus*)

Legal Status

State: None

Federal: None

Critical Habitat: N/A

Recovery Planning: N/A

Notes: There is disagreement about the taxonomic relationship of kit fox (*Vulpes macrotis*) and swift fox (*V. velox*) and subspecific designations for kit fox (e.g., Dragoo et al. 1990; Mercure et al. 1993; 57 FR 28167–28169).

Taxonomy

The kit fox (*V. macrotis*) is in the family Canidae and is the smallest canid species in North America (McGrew 1979). Descriptions of its physical characteristics can be found in McGrew (1979). While the desert kit fox (*V. m. arsipus*) is referred to in this profile as a subspecies of the kit fox, the taxonomy of this group has been uncertain and controversial, both at the species and subspecies levels. Dragoo et al. (1990) concluded that, based on genetic data, all arid-land foxes in North America pertained to a single species, the swift fox (*Vulpes velox*), and that morphometric data indicated that all kit foxes should be recognized as a subspecies of the swift fox. However, in a 90-day finding regarding a petition to remove the federally listed endangered San Joaquin kit fox (*V. macrotis mutica*) subspecies from the endangered species list based on the argument that the subspecies was not a valid taxon, the U.S. Fish and Wildlife Service (USFWS) asserted that the morphometric data presented by Dragoo et al. (1990) acknowledged the separation between the kit fox and swift fox (57 FR 28167–28169). The USFWS further cited a yet unpublished genetic study indicating that the mitochondrial DNA haplotype of the kit foxes and swift foxes was more geographically structured than that of larger canids and that gene flow between the two taxa was restricted (57 FR 28167–28169). The results of the genetic study cited in the 90-day finding were later published by Mercure et al. (1993), which supported the conclusion that kit fox and swift fox were

separate species. However, Mercure et al. (1993) also concluded, with the exception of the San Joaquin kit fox, that the genetic data did not support the other 10 subspecific designations of kit fox, including desert kit fox. Currently, no subspecies of kit fox are recognized, including desert kit fox and San Joaquin kit fox (Wilson and Reeder 2005). However, Mercure et al. (1993) acknowledged that the Colorado River may be a barrier to gene flow and that more extensive sampling would be needed to understand microgeographic barriers to gene flow such as the Colorado River.

Given that the desert kit fox subspecies is not listed as threatened or endangered, or otherwise has special state or federal status, these taxonomic issues are not relevant to its status as Covered Species under the Desert Renewable Energy Conservation Plan (DRECP).

Distribution

General

For the purpose of this profile, the range of the desert kit fox (*V. m. arsipus*) as described by Hall (1981) for *V. velox arsipus* is used. The desert kit fox is a year-round resident of the southwestern deserts of California, southern Nevada, the lower elevations of western and southern Arizona, and northern Mexico. Its western boundary that separates it from the federally listed and isolated San Joaquin kit fox subspecies is the Antelope valley in the west Mojave. The Tehachapi and Southern Sierra Mountain ranges form a physical barrier between desert kit fox and San Joaquin kit fox, although Mercure et al. (1993) suggest that the lower elevation Tehachapi range may be more permeable to movement than the Southern Sierra range.

Distribution and Occurrences within the Plan Area

Historical

The desert kit fox's range historically included the entire Plan Area.

Recent

There is a general lack of recent distribution information for this species; however, the desert kit fox's current distribution is

considered to include the entire Plan Area. Figure SP-M04 shows the Plan Area.

Natural History

Habitat Requirements

Kit foxes generally inhabit arid regions that receive less than about 16 inches (400 millimeters) of rain annually (Tannerfeldt et al. 2003). In the Plan Area, desert kit fox primarily occurs in open desert scrub habitats on gentle slopes. Creosote bush scrub in California is the most common habitat association for desert kit fox in California (McGrew 1979). A similar association with creosote brush scrub for den sites has been documented in Arizona (Zoellick 1985; Zoellick et al. 1989). In the Great Basin Desert portion of the Plan Area, suitable habitat includes saltbush (*Atriplex* spp.) scrubs. Penrod et al. (2012) created a suitable habitat model for desert kit fox that covers the Plan Area and that incorporates vegetation, topography, and road density and classifies habitat as good, fair, marginal, and unsuitable. “Good” habitat includes creosote bush–white bursage desert scrub or mixed salt desert scrub on slopes less than 5% and with low road density. “Fair” habitat includes areas with slopes less than 5% and other vegetation types suitable for kit fox such as playas and washes or medium road densities. “Marginal” habitat includes areas with slopes of 5%–15% or vegetation/cover types marginal for kit fox such as dune fields. “Unsuitable” areas includes slopes greater than 15%, unsuitable vegetation/cover types such as unvegetated lands, rocklands, bedrock, cliff and outcrop, and developed and cultivated lands.

O’Farrell and Gilbertson (1986) documented desert kit foxes in the western Mojave Desert northeast of California City and south of the El Paso Mountains (Rand Open Area and Desert Tortoise Research Natural Area) using habitat dominated by *Larrea-Schismus-Erodium*, with relatively low cover of burro bush (*Ambrosia dumosa*). O’Farrell and Gilbertson (1986) characterized the study sites as disturbed by sheep grazing and off-highway vehicles (OHVs). Similarly, kit foxes in western Arizona were observed to den in creosote scrub and spend more time in creosote scrub than expected based on its availability relative to other habitat types (Zoellick et al. 1989). About 80% of kit fox dens in the Great Basin Desert in western Utah were in sparsely

vegetated shadscale flats with low vegetation of 8–10 inches, and with shadscale (*Atriplex confertifolia*) as the most common species (Egoscue 1956). Egoscue (1956) noted that while dens were located in areas with low vegetation and high visibility, prey productivity was low in these areas, requiring individuals to travel more than a mile to forage in more productive habitats. However, Arjo et al. (2003) discuss a potential tradeoff of vegetation structure around dens, with lower vegetation height providing better detection of advancing predators and higher vegetation height providing better concealment and possibly higher invertebrate prey availability. Proximity of water does not appear to be a factor in kit fox den selection (Egoscue 1956), and the species can meet its water needs through prey (McGrew 1979).

Dens are an important resource for kit fox because they provide microclimate moderation and protection from predators, and may be a limiting resource for kit fox distribution (Arjo et al. 2003). Kit foxes form monogamous pairs (at least through a breeding season) and often small family groups that occupy den complexes (Ralls and White 2003; Ralls et al. 2007). Kit foxes may dig their own dens, use dens created by other species such as badger (*Taxidea taxus*), or expand on burrows created by smaller species such as kangaroo rats (*Dipodomys* spp.) and prairie dogs (*Cynomys* spp.) (Arjo et al. 2003; Tannerfeldt et al. 2003). Whether kit foxes dig their own dens or use dens and burrows created by other species may depend on the availability of preexisting dens/burrows, with kit foxes rarely digging dens when they do not have to (Tannerfeldt et al. 2003). Desert kit fox dens in the western Mojave in the O'Farrell and Gilbertson (1986) study tended to be on west- and northwest-facing slopes on friable soils with an absence of stones, caliche, or hardpan (O'Farrell and Gilbertson 1986). Kit foxes may also occasionally den in manmade culverts (Egoscue 1956; O'Farrell and Gilbertson 1986). Arjo et al. (2003) discuss the hypothesis that the orientation of natal den entrances may be related to protection from prevailing winds and provide other microclimatic advantages, suggesting that entrance orientation may be related to local climatic factors. Selection of den sites may also depend on the distribution of coyotes (*Canis latrans*), which is a common natural predator of kit foxes (e.g., Rall and White 1995; White et al. 1995; White and Garrott 1997; Kozłowski et al. 2008) and direct competitor for resources (White et al. 1995; Arjo et al. 2003, 2007; Kozłowski et al. 2008). For example, in western Utah, kit foxes may have altered

their distribution and den sites to more mountainous areas and areas vegetated by non-native grasses in response to increased coyote populations in the study area since 1959 (Arjo et al. 2003) (see discussion in Ecological Relationships).

Kit fox dens typically have multiple entrances (Egoscue 1956; O'Farrell and Gilbertson 1986; Tannerfeldt et al. 2003). In the O'Farrell and Gilbertson (1986) study, dens averaged 3–5 entrances, with up to 10 entrances. Natal (pupping) dens used by desert kit foxes from January to the end of May were larger and had more entrances (5–8) than non-natal dens (3–4) used from June through December (O'Farrell and Gilbertson 1986), which also appears generally common in kit foxes (e.g., Arjo et al. 2003; Tannerfeldt et al. 2003).

Kit foxes use numerous dens, switching dens frequently, and dens tend to be clustered (Tannerfeldt et al. 2003). Clusters include several dens (in one study, up to 17) that may be more than 328 feet (100 meters) apart (Tannerfeldt et al. 2003). In San Joaquin kit fox, den switching may occur several times monthly and most often during the dispersal season, but switching is also related to age class with adults tending to use more dens than juveniles (Tannerfeldt et al. 2003). Although dens may be shared by pair-mates throughout the year, den sharing may be seasonally variable, with higher rates during December during the breeding season and lower rates in February when very young pups were present, for example (Ralls et al. 2007).

Natal dens in the western Mojave appeared to be spaced, with possible territorial exclusivity, with a minimum inter-den distance of approximately 1.25 miles (2 kilometers) (O'Farrell and Gilbertson 1986). This spacing may reflect territorial requirements and carrying capacity (O'Farrell and Gilbertson 1986). Similarly, in western Utah natal dens were at least 2 miles (3.2 kilometers) apart (Egoscue 1975). In San Joaquin kit fox, territories of adjacent social groups had only slight overlap (White and Ralls 1993).

Selection of den sites does not appear to be strongly related to nearby human activities, nor do kit foxes appear to actively avoid man-made features such as roads and structures. O'Farrell and Gilbertson (1986) found that most desert kit fox dens were within 492–656 feet (150–200 meters) of roads or trails in the western Mojave. Bjurlin et al. (2005) found that almost 10% of San Joaquin kit dens in the

Bakersfield area were within 100 feet of road centerlines and that some dens used features of major roads, including culverts, embankments and underpasses, and drainage basins or canals immediately adjacent to roads.

Foraging Requirements

Several studies in California, Arizona, and Utah, as summarized by Tannerfeldt et al. (2003), show that the primary food sources for kit foxes are rodents and lagomorphs, including jackrabbit (*Lepus* spp.) and cottontails (*Sylvilagus* spp.). Egoscue (1956) listed several prey species in the Great Basin Desert of western Utah, including black-tailed jackrabbit (*Lepus californicus*), kangaroo rat (*Dipodomys* spp.), and deer mouse (*Peromyscus maniculatus*), but also burrowing owl (*Athene cunicularia*), western meadowlark (*Sturnella neglecta*), horned lark (*Eremophila alpestris*) (notably all open ground-nesting species), side-blotched lizard (*Uta stansburiana*), and sand cricket (*Stenopelmatus* sp.). Similarly, on the Carrizo Plain in California, San Joaquin kit fox prey included kangaroo rats, pocket mice (*Chaetodipus* spp. and *Perognathus* spp.), deer mouse, black-tailed jackrabbit, desert cottontail (*Sylvilagus audubonii*), and California ground squirrel (*Spermophilus beecheyi*) (White and Ralls 1993). In the Plan Area, it is expected that primary prey for desert kit fox include black-tailed jackrabbit, desert cottontail, Merriam's kangaroo rat (*D. merriami*) (the most common and widespread kangaroo rat in the Plan Area), various pocket mice species, other rodents such as woodrats (*Neotoma* spp.) and California ground squirrel, and various small reptiles.

Hunting is almost strictly nocturnal, with kit foxes resting in their dens during the day (Egoscue 1956; White et al. 1995). As noted under spatial activity, individuals may move several miles daily, but it is likely that foraging distances are closely related to prey availability, which is likely variable spatially and temporally (Egoscue 1956).

Reproduction

The desert kit fox reproductive period in the Plan Area is generally December to late May (O'Farrell and Gilbertson 1986) (see Table 1), which is consistent with other parts of the kit fox's range (e.g., Egoscue 1956; McGrew 1979). In the O'Farrell and Gilbertson (1986)

study in the western Mojave, males maintained scrotal development throughout the year, but females were reproductive in December and January. Gestation is approximately 49–56 days (McGrew 1979), and females in the O'Farrell and Gilbertson (1986) study were lactating in March and April, indicating birth in February and March. Kit fox litters are 2–6 pups (Egoscue 1956; McGrew 1979; Tannerfeldt et al. 2003; USFWS 2010), and pups emerge from the natal den at about 4 weeks of age (USFWS 2010). Both adults provide care to pups. Initially males do most of the hunting while lactating females remain in the den (Egoscue 1956). In the O'Farrell and Gilbertson (1986) study, pups were absent from natal dens by the end of May. However, for San Joaquin kit fox, pups remain under the care of adults for 4 to 5 months, before beginning to disperse from their natal area as early as July and continuing through August and September (Moonjian 2007; USFWS 2010). Some offspring remain with their parents and help raise the next litter during the following year (USFWS 2010). Also in San Joaquin kit fox, yearling females may breed, with about 18% of monitored successfully reproducing (Cypher et al. 2000). Egoscue (1956) reported two lactating females in the same den on two occasions, with one instance apparently a mother and daughter.

Kit foxes generally exhibit monogamy, with pairs remaining together for several breeding seasons, and some pair bonds being permanent until the death of one of the pair (Egoscue 1956; O'Farrell and Gilbertson 1986; Ralls et al. 2007). In San Joaquin kit fox, Ralls et al. (2007) documented that 14 of 16 dissolutions of a pair were due to the death of a pair-mate, 1 was due to the male abandoning the female, and the other was due to a new male displacing the mate. Pair formation can occur throughout the year (Ralls et al. 2007).

Mortality rates in the O'Farrell and Gilbertson (1986) study were high with average observed longevity on the order of 10 months (range 8–14), although some individuals were still alive when the study was completed. Mortality resulted from several causes, including shooting, starvation, predation (likely coyote or dog), vehicle collisions, and den collapse (see Threats and Environmental Stressors). In a study of dispersal by San Joaquin kit fox, Koopman et al. (2000) found that more than 65% of dispersing juveniles died within 10 days of leaving their natal range. The primary cause of mortality of dispersing and

philopatric juveniles was predation. Kit foxes in zoos have lived 10–12 years (McGrew 1979), but such a long life span in the wild is unlikely.

Table 1. Key Seasonal Periods for Desert Kit Fox

	Jan	Feb	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	x	x	x	x							x
Dispersal						x	x	x			

Spatial Activity

Desert kit foxes are quite mobile and have relatively large home ranges. In the western Mojave, O'Farrell and Gilbertson (1986) estimated ranges of the approximately 494 acres based on radiotelemetry data. Data for other subspecies indicate at least as large to much larger home ranges, with home-range size likely related to resource availability. For San Joaquin kit fox, Koopman et al. (2001) determined a mean adult home-range size of approximately 1,072 acres and a mean pup home-range size of 325 acres on the Naval Petroleum Reserves in western Kern County (USFWS 2010). Briden et al. (1992, as cited in USFWS 2010) found that denning ranges (the area encompassing all known dens for an individual) for San Joaquin kit fox averaged approximately 1,169 acres in western Merced County. White and Ralls (1993) estimated a mean home range for San Joaquin kit fox of approximately 2,866 acres at the Carrizo Plain in 1990 and 1991, but noted these home ranges were large and likely reflected drought conditions and prey scarcity. Home ranges during this study were also relatively exclusive, with little overlap between individuals of the same sex (White and Ralls 1993). At the Camp Roberts Army National Guard Training Site in northern San Luis Obispo County, radiotelemetry documented mean home ranges for San Joaquin kit fox of approximately 5,782 acres (Root and Eliason 2001, as cited in USFWS 2010). White and Ralls (1993) suggested that large, exclusive home ranges during periods of drought may be an adaptation to episodic prey scarcity and a means to maintain their own body mass and condition.

Daily movements of desert kit foxes in western Arizona during the period of December through March averaged 8.9 miles (14.3 ± 0.71

kilometers/night) for males and 7.4 miles (11.8 ± 1.08 kilometers/night) for females (Zoellick et al. 1989). Males tended to move greater distances during the breeding season compared to pup rearing and pair formation periods (Zoellick et al. 1989). O'Farrell and Gilbertson (1986) did not observe young remaining in their natal territory and recorded a maximum dispersal of approximately 20 miles (32 kilometers) by a female. Egoscue (1956) reported movements up to 20 miles by juvenile kit foxes in western Utah. However, in the San Joaquin kit fox, which has been much more extensively studied than desert kit fox in the Plan Area, some offspring remain with their parents (Ralls et al 2001). Young of this subspecies may also remain their natal territory. In one study spanning 16 years, 33% of tracked juveniles dispersed from their natal territory, with significantly more males dispersing than females, and the average dispersal distance was 4.8 miles (range of 1.1 to 20 miles) (Koopman et al. 2000). Most dispersal occurred in the first year of the animal's life. Briden et al. (1992, as cited in USFWS 2010) documented dispersals of 1.2 to 12 miles. Four long-distance dispersals of between 25 and 50 miles were documented between Camp Roberts and Fort Hunter Liggett Military Reserve in Monterey County and the Carrizo Plain (California Air National Guard 2008, as cited in USFWS 2010).

Koopman et al. (2000) did not find any significant relationships between dispersal patterns in San Joaquin kit fox and demographic factors, including population density, the number or sex ratio of adults, the sex ratios of juveniles, or the proportion of new juveniles in the population. They also did not find a relationship with ecological factors, including leporid density and total prey density, small mammal abundance, or coyote abundance.

Whether the spatial activity patterns exhibited by San Joaquin kit fox are applicable to desert kit fox is unknown, but it is likely that spatial activity by desert kit fox (i.e., territory and home range use, spacing, dispersal, etc.) is also dynamic and potentially related to demographic and environmental factors such as prey availability (see discussion in Ecological Relationships).

Ecological Relationships

Fairly extensive research has been conducted on the ecological relationships of kit foxes to other species, and in particular to coyotes, which is a common predator of kit foxes (e.g., Rall and White 1995;

White et al. 1995; White and Garrott 1997; Kozlowski et al. 2008) and direct competitor for prey (e.g., White et al. 1994, 1995; Arjo et al. 2007; Kozlowski et al. 2008). A brief summary of some of these studies, as they may relate to conservation of the desert kit fox in the Plan Area, is provided here.

Several studies have noted dramatic kit fox population fluctuations in relation to prey availability. For example, in San Joaquin kit fox, Cypher et al. (2000) found that high kangaroo rat densities positively influenced the growth of a kit fox population, while Moonjian (2007) found that low densities of kit foxes in the Palo Prieto area of western Kern County were associated with low densities of kangaroo rats. Local extirpations have also been linked to the previous loss of kangaroo rat populations (Cypher et al. 2000). White and Ralls (1993) found that prey scarcity related to drought reduced reproductive success in San Joaquin kit fox on the Carrizo Plain, with no reproduction by nine tracked females in 1990.

Prey selection by San Joaquin kit fox may also track availability. A 15-year study at the Naval Petroleum Reserves in western Kern County found that the dominant prey item alternated over time between kangaroo rats and leporids (Cypher et al. 2000). Similar prey studies have not been conducted for desert kit fox, but it is expected that patterns would be similar because desert rodent and lagomorph populations also vary substantially in relation to environmental conditions and possibly demographic factors. For example, Beatley (1969) found that desert rodent reproduction and population densities in southern Nevada were strongly associated with fall rain and production of winter annuals plants. Black-tailed jackrabbit densities and distribution appear to have a more complex relationship with environmental conditions because their diet shifts between seasons, locations, years, and vegetation types (Hayden 1966; Johnson and Anderson 1984; Wansi et al. 1992). The length of the jackrabbit breeding season appears to be related to the production of herbaceous vegetation (Lechleitner 1959), and reproductive activity appears to be density-dependent, which can result in wide population fluctuations on 7–10-year cycles (French et al. 1965; Wagner and Stoddart 1972; Smith 1990).

Home-range size also appears to vary in relation to prey availability, with smaller home ranges where lagomorphs are abundant and larger

home ranges when desert kit foxes have to rely on small prey such as kangaroo rats and other small rodents (Zoellick and Smith 1992).

Coyotes are both predators of kit foxes and direct competitors for food, with substantial spatial, temporal, and dietary overlap (White et al. 1994, 1995; Kozlowski et al. 2008). Habitat and land use changes that attract coyotes therefore would likely have an adverse effect on desert kit foxes. Arjo et al. (2007), for example, suggest that invasion of a site in western Utah (the same site studied by Egoscue in the 1950s) by cheatgrass (*Bromus tectorum*), replacing native Great Basin shrub communities, and the addition of artificial water sources have altered prey abundance and attracted coyotes, to the detriment of kit foxes. Kit foxes do not require free water and are less water-limited than coyotes. The increased abundance of coyotes may have increased direct competition for food resources, with kit foxes having to focus on small rodents due to increased predation of lagomorphs by coyotes (Arjo et al. 2007). On the same Utah site, Kozlowski et al. (2008) found that kit foxes and coyotes used space within their home ranges differently, with kit foxes using areas of vegetation and ruggedness not favored by coyotes, but interactions were still common and 56% of kit fox mortalities were attributed to coyotes.

Population Status and Trends

Global: Apparently Secure (NatureServe 2012)

State: Not ranked

Within Plan Area: Not ranked

The desert kit fox currently does not have federal or California special status, although it is protected from hunting as a fur-bearing mammal under Title 14 of the California Code of Regulations, Section 460. Population status and trends in the Plan Area are unknown, but it has been characterized as uncommon to rare in arid regions in California (Zeiner et al. 1990). Meany et al. (2006) state that kit fox populations “plummeted” in the last half of the 19th and early 20th century due to predator and rodent controls. They report that the kit fox population in Colorado may be close to extirpation, populations in Oregon and Idaho are extremely low, and populations in the Great Basin Desert in Nevada and Utah may be in decline. The only states Meaney et al.

(2006) indicate may still have stable populations are Arizona, New Mexico, and Texas.

In March 2013 The Center for Biological Diversity (CBD) submitted a petition to the California Department of Fish and Wildlife (CDFW) to list the desert kit fox as threatened under the California Endangered Species Act (Kadaba et al. 2013). The CBD cited large-scale energy development as a primary threat, in concert with OHV use, grazing, agriculture, military activities, urbanization, climate change, and increased anthropogenic disease risks (Kadaba et al. 2013). Although the species' status and trends in the Plan Area are unknown, it is reasonable to assume that the threats and stressors cited in the CBD petition have resulted in loss, fragmentation, and degradation of habitat for kit fox in the Plan Area and at least local impacts on local populations subject to these threats and stressors (see Threats and Environmental Stressors). Whether these effects, as outlined in the petition, have risen to the level of warranting a listing as threatened is yet unknown and await analysis and determination by CDFW.

Threats and Environmental Stressors

An initial cause of population declines in kit fox was predator and rodent controls in the 19th and 20th centuries (Meaney et al. 2006). Several threat factors cited by Meaney et al. (2006) for Colorado that may apply to the desert kit fox in the Plan Area are habitat degradation, loss and fragmentation from development, roads, recreation, and grazing. The expansion and increased abundance of coyotes, which is the main predator of kit foxes, is also a threat.

A potentially devastating current threat to desert kit fox is canine distemper, which was determined to be the cause of death of several kit foxes at and near a solar energy project located west of Blythe in fall 2011 (Clifford et al. 2013). The source of the distemper outbreak is not known and may have been a domestic dog or native wildlife such as badger. This distemper outbreak is the first documented incident in wild kit foxes (Clifford et al. 2013). Subsequent trapping of 39 individuals in January 2012 at the outbreak site found that all appeared healthy, but the capture rate at the affected site was low, indicating a reduction in the local population (Clifford et al. 2013). Although the recent outbreak of canine distemper is the first documented incident in wild kit foxes, O'Farrell and Gilbertson (1986)

suggested that canine distemper or some other viral or bacterial disease may have been a causal factor in the apparent starvation deaths of several desert kit fox individuals during a study conducted from 1977 to 1979 in the western Mojave, because one clinical symptom of distemper is anorexia and gradual loss of activities, which can result in starvation. O'Farrell and Gilbertson (1986) observed that the animals died over a short time period, died underground, were emaciated and had no food in their gastrointestinal tracts, showed evidence of diarrhea, and had conjunctival secretions. Unfortunately the individuals were recovered too late for histopathological diagnosis (O'Farrell and Gilbertson 1986).

In addition to habitat impacts and disease, it is expected that desert kit fox is also vulnerable to various human activities, including recreation such as OHVs. However, O'Farrell and Gilbertson (1986) found that most dens were within 490–656 feet (150–200 meters) of roads or jeep trails in the Rand Open Area in the western Mojave that was subject to unlimited OHV activity during the study from 1977 to 1979 (i.e., there was no apparent tendency to locate dens away from roads or trails). However, mortalities related to shooting, vehicle collisions, den collapse (which could result from OHV activity), and potentially canine distemper (which could be transmitted by dogs) were observed.

In more urbanized areas, vehicle collisions are a frequent source of mortality of kit foxes. Bjurlin et al. (2005) found that vehicle collisions were the primary cause of mortality of San Joaquin kit foxes in the Bakersfield area, whereas predation is the more common cause of mortality of the subspecies in natural areas (e.g., Ralls and White 1995). Bjurlin et al. (2005) found that while kit foxes frequently crossed local roads, collisions were statistically more likely to occur on arterials with higher traffic densities and speeds; about 69% of all documented strikes were on four- and six-lane arterials and about 88% of all strikes were on roads with posted speed limits of 45, 50, or 55 miles per hour (56% of strikes were on roads with a 55-mile-per-hour speed limit). Bjurlin et al. (2005) also found that collisions on roads were disproportionate to males during the winter in association with territorial defense, mating, and exploratory movements. Further, even though den selection was not related to road proximity, close proximity of dens to roads increased collision risk.

Desert kit fox is also vulnerable to rodenticide poisoning (Shitoskey 1975; Meaney et al. 2006). Shitoskey (1975) demonstrated that three rodenticides—sodium monofluoroacetate (compound 1080), strychnine alkaloid, and zinc phosphate—were lethal to kit fox when administered directly. Sodium monofluoroacetate and strychnine alkaloid were also lethal when kit fox ingested kangaroo rats killed by the two rodenticides, but kit fox was able to tolerate kangaroo rats contaminated with zinc phosphate.

Military training will be an ongoing activity in the Plan Area, and noise associated with such activities, including from aircraft, may be a concern for overall stability of the desert kit fox, including potential direct effects on kit foxes and indirectly through effects on prey abundance and availability. Bowles et al. (1995) examined the effects of aircraft noise on kit fox and the desert rodent community on the Barry M. Goldwater Air Force Range in Arizona from 1991 to 1994. Monitoring on affected and control sites revealed no large differences in kit fox or rodent communities that could be attributed to aircraft noise, and observed differences between exposed and control population generally were within those expected through natural variability. Survival (as measured by “days known alive”) for kit foxes on control and exposed sites were not significantly different, and the median survival days was actually higher on the exposed site at 223 days vs. 209 days for the control site. Individual weights (a measure of physical condition) and home-range sizes were also not different for the control and exposed sites. For the desert rodents, no statistical differences were found for species diversity, population densities, and weights (a measure of physical condition) between control and exposed sites. Annual rodent survival rates were higher in control sites, and recruitment was higher on exposed sites.

Conservation and Management Activities

The desert kit fox is not a special-status species, nor is it covered under any existing conservation plans in or adjacent to the Plan Area. It is not explicitly addressed in federal land use planning, such as the Bureau of Land Management’s (BLM’s) West Mojave Plan (2005), Northern and Eastern Colorado Plan (2002a), and Northern and Eastern Mojave Plan (2002b). It is also not explicitly addressed by the

National Park Service general management plans for Mojave National Preserve, Death Valley National Park, and Joshua Tree National Park.

A management and monitoring plan for desert kit fox was developed for the Genesis Solar Energy Project (AECOM 2012) where several mortalities attributed to canine distemper occurred. The plan includes several avoidance and minimization measures for the project: pre-construction surveys; den classification and excavation of inactive den complexes in the construction area to prevent reuse; monitoring of potential and known active den complexes; exclusion of kit foxes from den complexes using passive methods; and protocols for handling sick, injured, or dead kit foxes.

Data Characterization

There is a lack of population and distributional information for desert kit fox in the Plan Area, including use of and movement through landscape. The local ecology of the species and the San Joaquin kit fox subspecies is well studied in some locales (e.g., western Utah, western Arizona, central California) with regard to life history traits and ecology, but only one older ecological study for the desert kit fox in the western Mojave portion of the Plan Area has been conducted (i.e., O'Farrell and Gilbertson 1986).

Management and Monitoring Considerations

Because suitable den sites may be a limiting resource for desert kit fox, maintaining suitable denning habitat may be important for conservation of the species, including relatively open habitat, gentle slopes, and friable soils (O'Farrell and Gilbertson 1986; Arjo et al. 2003). Other important factors may be conversion of habitats to annual grassland that could affect prey abundance and provision of the artificial water sources that could attract coyotes that are predators of kit foxes and direct competitors for resources. The ability of kit foxes to move through the landscape may be enhanced by providing culverts in key locations.

Understanding causes of death is also an important management and monitoring consideration for desert kit fox, especially those with potential anthropogenic causes or interactions, including diseases

such as canine distemper, vehicle collisions, and coyote predation and competition.

General ecological and behavioral studies for desert kit fox are also lacking for the Plan Area. Studies of other kit fox populations across the southwest reveal substantial variability in various life history traits, including habitat selection, demographics, predator-prey relationships, and vulnerability to various threats and stressors, suggesting that effective conservation and management of the desert kit fox in the Plan Area will require additional Plan-specific information.

In addition to maintaining suitable habitat and prey availability, mobility across the landscape is an important management and monitoring consideration, especially across roads that can be significant contributors to mortality. Kit foxes are known to cross highways at grade, but their use of below-grade crossings (e.g., culverts, bridges, and underpasses) is less understood. Boarman and Sazaki (1996) incidentally documented desert kit foxes activity at culverts under State Route (SR) 58 in the Plan Area approximately 7 miles east of Kramer Junction during a study of desert tortoise (*Gopherus agassizii*). The study observed kit fox activity around culverts, including steel pipes that were 2.9–4.9 feet (0.9–1.5 meters) in diameter, concrete pipes 55 inches (1.4 meters) in diameter, and concrete boxes 9.8–11.8 feet (3–3.6 meters) wide by 5.9–9.8 feet (1.8–3 meters) high, but it did not provide data documenting actual crossings using the culverts or whether culverts of certain dimensions were used or avoided. Due to telemetry equipment failures and low capture rates, a recent study of below-grade crossings of the four-lane SR-58 west of Barstow by desert kit foxes by Clevenger et al. (2010) was generally unsuccessful in documenting whether kit foxes cross the highway using available corrugated metal culverts, cement box culverts, and bridge crossings. Two individuals were documented successfully crossing the highway, but it is unknown whether the crossings were through below-grade structures or at grade across the highway. However, Clevenger et al. (2010) did document two apparent swift fox crossings of Interstate 70 in Colorado using reinforced concrete pipe culverts and several crossings of Interstate 90 in South Dakota using culverts, including at least four two-lane and one four-lane crossing, and possibly a six-lane crossing. The dimension of the box culvert in the four-lane crossing was relatively tall and wide (84 x 84 inches) (Clevenger et al. 2010). A

recent camera monitoring study for the Coachella Valley Multiple Species Habitat Conservation Plan (MSHCP) conducted at six highway underpasses in the Coachella Valley from September 2011 to April 2012 failed to detect any desert kit foxes (Murphy and Barrows 2012). However, the status and distribution of kit fox in the MSHCP plan area is unknown, so its apparent absence at the monitored underpasses is difficult to interpret.

Kit foxes in urbanized areas are known to cross roads, including six-lane arterials, but the risk of vehicle collisions is high on four- and six-lane arterials and was found to be the main cause of mortality in the Bakersfield area (Bjurlin et al. 2005). Bremner-Harrison et al. (2005) conducted a 1-year study of road culvert use in Kern County along Interstate 5, SR-14, and SR-58 and failed to document any use of culverts to cross roads. They hypothesized that kit foxes may associate the closed spaces of culverts with increased predation risk from coyotes, dogs, and bobcats (*Lynx rufus*) that were detected in and around crossing structures. They did not study use of large structures for crossing such as bridges over larger washes, and kit fox use of large structures is unknown. The diverse desert terrain in the Plan Area includes many culvert crossings under existing roads for drainage, but use of these culverts by desert kit fox is unknown (e.g., Clevenger et al. 2010), although O'Farrell and Gilbertson (1986) documented use of a road culvert as a den.

Bremner-Harrison et al. (2005) made several recommendations regarding road crossings for kit fox that may be applicable to the Plan Area:

1. Conduct further field investigations to determine whether kit foxes are indeed avoiding structures and crossing roads, or are generally avoiding roads.
2. If opportunities arise, repeat this investigation in areas with median barriers to determine whether kit foxes are more likely to use crossing structures in such areas or simply abandon attempts to cross roads.

3. In areas where median barriers are present along highways, recommendations to reduce adverse impacts to kit foxes include:
 - a. install fencing to exclude kit foxes from the highway and direct them to crossing structures;
 - b. design crossing structures to accommodate use by the largest animal species occurring in the local ecosystem, and
 - c. place artificial dens within crossing structures and near entrances to provide escape cover for kit foxes. (Bremner-Harrison et al. 2005, p. 42)

Based on other known and likely threats and stressors to kit fox in the Plan Area, other management and monitoring considerations include:

- Developing demographic data for desert kit fox in the Plan Area, including population size and distribution
- Understanding the ecological relationships between kit foxes and coyotes in the Plan Area
- Understanding predator/prey relationships and maintaining and enhancing prey populations in areas supporting kit foxes
- Managing the use of rodenticides and other pesticides
- Managing and monitoring the incidence of diseases such as canine distemper.

Predicted Species Distribution in Plan Area

This section provides the results of habitat modeling for desert kit fox, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 15,686,640 acres of modeled suitable habitat for desert kit fox in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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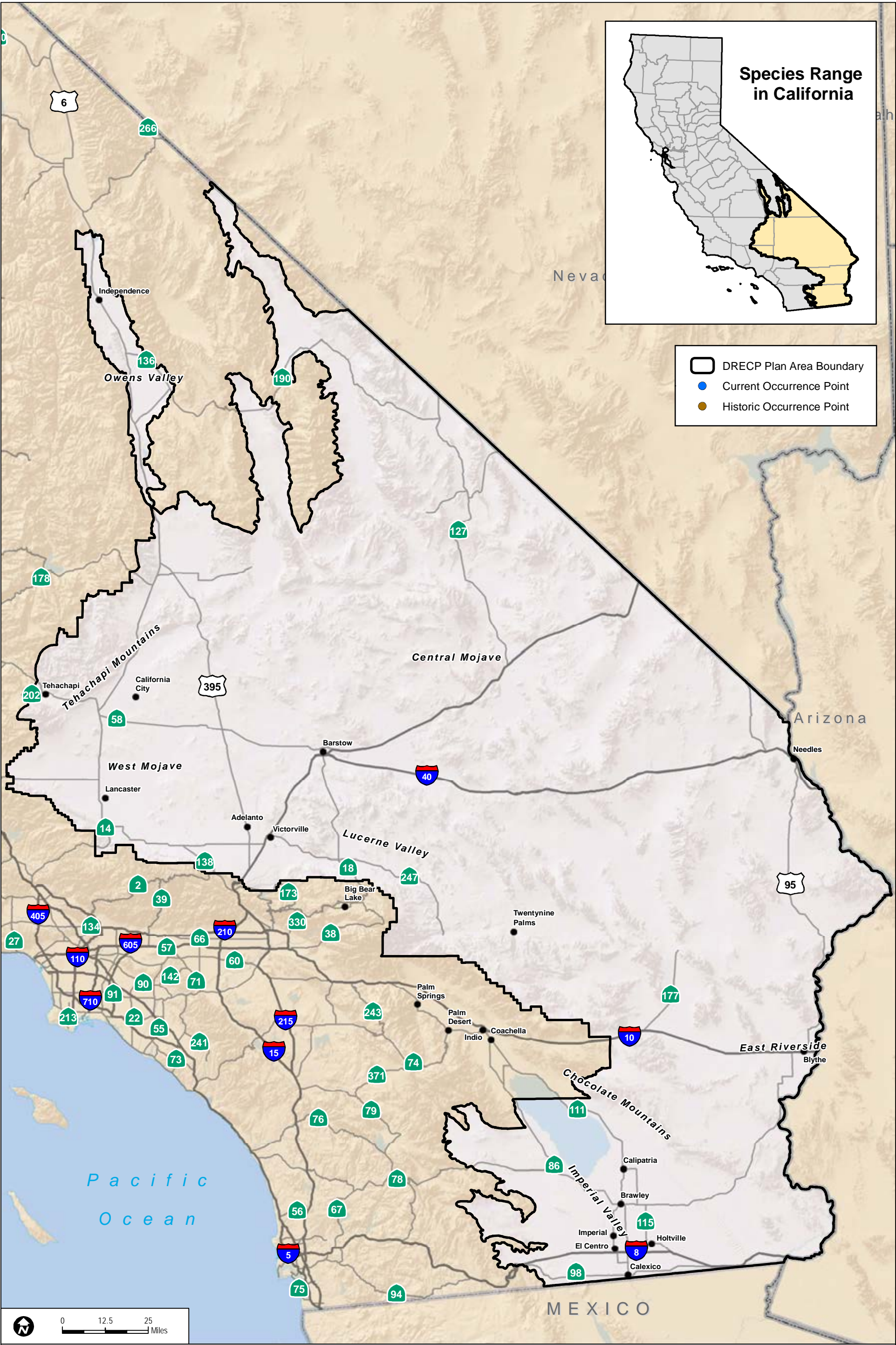
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M04
Desert Kit Fox Occurrences in the Plan Area

Mohave Ground Squirrel

(*Xerospermophilus mohavensis*)

Legal Status

State: Threatened

Federal: None

Critical Habitat: N/A

Recovery Planning: No formal state or federal recovery plans have been prepared.

Note: The U.S. Fish and Wildlife Service (USFWS) published a 12-month finding on October 6, 2011, that listing of the Mohave ground squirrel (*Xerospermophilus mohavensis*) is not warranted at this time (76 FR 62214–62258).



Photo courtesy of Phil Leitner.

Taxonomy

The Mohave ground squirrel (*Xerospermophilus mohavensis*) was discovered by F. Stephens in 1886 and described as a distinct monotypic species by Merriam in 1889. The type locality is the Mohave Desert near Rabbit Springs, about 24 kilometers (15 miles) east of Hesperia in San Bernardino County (Helgen et al. 2009).

The Mohave ground squirrel belongs to the family Sciuridae, which includes rodents that dig their own burrows (Gustafson 1993). Previously recognized as *Spermophilus mohavensis*, based on a review of morphometrics (measurement of external form and structure) and molecular phylogenetics (evolutionary relationships within and between groups), the Mohave ground squirrel is now recognized as *Xerospermophilus mohavensis* (Helgen et al. 2009). The Mohave ground squirrel is a distinct, full species with no recognized subspecies (Helgen et al. 2009). However, there has been some question about the recognition of the round-tailed ground squirrel (*Xerospermophilus tereticaudus*) and the Mohave ground squirrel as distinct species (Gustafson 1993; Hafner 1992; Hafner and Yates 1983). The two squirrels are closely related and have a contiguous, but not overlapping, geographic range (Best 1995; Hafner 1992).

Hafner and Yates (1983) described a narrow hybridization zone in the ranges of the two species in an area northwest of Helendale and near Coyote Dry Lake northeast of Barstow, but studies by Hafner and Yates (1983) and Hafner (1992) demonstrated that there were sufficient chromosomal, genetic, morphological, and ecological differences to warrant distinct species recognition.

Distribution

General

Endemic to California, the Mohave ground squirrel is exclusively found in the northwestern Mojave Desert in San Bernardino, Los Angeles, Kern, and Inyo counties (Best 1995; Figure SP-M05).

Distribution and Occurrences within the Plan Area

Historical

The presumed historical range of the Mohave ground squirrel within the northwestern Mojave Desert was bounded on the south and west by the San Gabriel, Tehachapi, and Sierra Nevada mountain ranges; on the northwest by Owens Lake, and on the northeast by the Granite and Avawatz mountains; and on the east and southeast by the Mojave River (Leitner 2008; MGSWG 2011). In addition, the species was historically found in one locality east of the Mojave River in the Lucerne Valley. Its historic range covered about 20,000 square kilometers (km²) (7,722 square miles [mi²]) (Gustafson 1993), which is the smallest geographic range of any ground squirrel species in the United States. However, for the 12-month finding for the species published in October 2011, USFWS used a somewhat larger historical range of approximately 21,525 km² (8,311 mi²) (76 FR 62214–62258). USFWS also stated in the 12-month finding that the range of the Mohave ground squirrel may be larger than defined in the finding or previously published based on recent sightings such as in an interior valley of the Tehachapi Mountains and in the Panamint Valley about 8 kilometers (5 miles) north of the defined range (76 FR 62214–62258).

Based on the range used by Leitner (2008), about 88% of the historical range of the species is within the Plan Area (only the Coso Range in the northern extent of its historic range is excluded).

Prior to conversion of native desert habitats in the Antelope Valley west of Palmdale and Lancaster to agriculture and residential and commercial development, there was potential habitat for the Mohave ground squirrel, but there are no historical or recent occurrence records in this area west of State Route 14 (Leitner, pers. comm. 2012).

Approximately 28% of the California Natural Diversity Database (CNDDB) records for the Mohave ground squirrel are historical or have no date. These records are located throughout the species' range (Figure SP-M05) (CDFW 2013).

Recent

The current range may be reduced from the historical range as a result of the possible extirpation of the Mohave ground squirrel in the western portion of the Antelope Valley; although there is suitable desert scrub, there are no historical records for areas west of State Route 14. The species has been extirpated from much of the Victorville area due to agricultural and more recent rapid urban development, but there are a few recent CNDDB records, including from 2005, 2007, and 2011, for the Adelanto area (CDFW 2013; Dudek 2013; Figure SP-M05), indicating a possible relict population in the southern portion of its range (Leitner, pers. comm. 2012).

Habitat for the species has been reduced by development of agricultural uses, grazing, urbanization, military activities, energy production, and recreation (MGSWG 2011). The current occupied range is estimated to be about 19,000 km² (6,640 mi²) (MGSWG 2011).

The occurrence of Mohave ground squirrel is likely to be patchy within its range, even within apparently suitable habitat (MGSWG 2011). However, as noted by Leitner (2008), occurrence records tend to be concentrated in certain areas where trapping studies have been focused; these studies are discussed in more detail below. There has not been a systematic, range-wide census or statistically based random sampling study to determine occupation throughout the species' range (Leitner 2008). About 88% of the geographic area of

known existing populations of the species, based on Leitner (2008), occur in the Plan Area (only a portion of the Coso Range-Olancho Core population is outside this area).

Recent (after 1990) records from the CNDDDB and West Mojave Plan Mohave ground squirrel transect data and other California Department of Fish and Wildlife (CDFW) data include location occurrences ranging from Inyo County in the north to 3 miles southwest of Rabbit Lake in the south. The eastern extent ranges to the Granite Mountains and Fort Irwin and the westernmost record is just east of Oak Creek (Figure SP-M05) (Dudek 2013).

Leitner (2008) provides the most current status of the Mohave ground squirrel based on compilation of a database, including unpublished field studies, surveys, and incidental observations for the 10-year period from 1998 through 2007 (Table 1). This database includes 1,140 trapping sessions, of which 102 resulted in observation of the species, and 96 additional incidental observations. Most of these studies and observations have been conducted in the southern part of the species' range south of State Route 58 and no range-wide systematic or statistically based random sampling has been conducted to characterize the species' status throughout its range. Leitner (2008) emphasizes that there are large areas of potential habitat where the species' status is unknown, especially on the China Lake Naval Air Weapons Station and Fort Irwin.

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
Inyo County between Olancho and Haiwee Reservoir, Coso Range within China Lake Naval Air Weapons Station	Detected on five trapping grids, including Lee Flat just inside Death Valley and the northernmost occurrence record. Four other incidental records, including in north Panamint Valley several kilometers north of generally accepted range.
Ridgecrest area	Detected on 5 of 10 trapping grids in vicinity of Ridgecrest and 6 of 10 grids along State Route 176 east of Ridgecrest. No individuals trapped at two sites in Spangler Hills southeast of Ridgecrest.

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
Little Dixie Wash extending from Inyokern southwest to Red Rock Canyon State Park	Detected on 6 of 7 trapping grids scattered throughout valley and more than 20 incidental observations. Species widespread in area.
Fremont Valley to Edwards Air Force Base	No detections in last 10 years on 6 trapping grids in Fremont Valley. Thirteen records around periphery of Desert Tortoise Natural Area (DTNA) and likely to be present within DTNA. Two incidental records northeast of town of Mojave, but protocol trapping studies in area have been negative. Ten trapping and incidental observation records for area north of Boron and Kramer Junction. Species likely widespread across region.
Wind farm southwest of Mojave (outside accepted range but appears to have suitable habitat)	No detections at 24 trapping grids southwest of town of Mojave. Two unconfirmed observations in CNDDDB.
Edwards Air Force Base	Extensive monitoring conducted, with 6 observations on 40 trapping grids from 2003–2007. Distribution of species on Edwards Air Force Base is well documented.
Los Angeles County desert area	No detections on 52 trapping grids. Four positive records in small area near Rogers Dry Lake on Edwards Air Force Base.
Victor Valley to Barstow	Extensive surveys of Adelanto and western Victorville area with two trapping records and one incidental observation. One capture near intersection of U.S. 395 and I-15. These records indicate small residual population in area. No records east of Mojave River since 1955, but not well sampled in last 10 years. No detections on three trapping sites from El Mirage Dry Lake north and east toward Barstow.
Barstow area	Three records – one record about 3.5 miles south of Barstow near landfill and outside accepted range and two

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
	records west of City. One detected at the edge of alfalfa field near Harper Dry Lake and the other trapped about 6.1 miles west of Hinkley near State Route 58.
Coolgardie Mesa and Superior Valley north of Barstow	Positive records for three trapping grids and at least seven incidental observations.
Pilot Knob area	Detected five sites from Cuddeback Dry Lake east to the boundary of the China Lake Naval Air Weapons Station.

Source: Leitner 2008.

Approximately 52% of the CNDDDB records are located on public lands managed by the BLM, Department of Defense, California Department of Transportation, Department of Parks and Recreation, Kern and San Bernardino counties, and the Los Angeles Department of Water and Power). Approximately 21% are located on privately owned lands. The ownership of the remaining 27% of the CNDDDB records is unknown (CDFW 2013).

Natural History

Habitat Requirements

The Mohave ground squirrel occurs in a variety of desert shrubland habitats (Table 2). Although most often found in creosote bush scrub, it has also been recorded in desert saltbush scrub, desert sink scrub, desert greasewood scrub, shadscale scrub, Joshua tree woodland, and Mojave mixed woody scrub (Best 1995; 75 FR 22063–22070; MGSWG 2011). Mohave ground squirrel typically occupies areas with open vegetative cover and small bushes (< 0.6 meter (2 feet) in height) spaced approximately 6 to 9 meters (20 to 30 feet) apart (Best 1995).

Table 2. Habitat Associations for Mohave Ground Squirrel

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Creosote bush scrub, Desert saltbush scrub, Desert sink scrub, Desert greasewood scrub, Shadscale scrub, Joshua Tree woodland, Mojave mixed woody scrub	Primary habitat	Active and Inactive Season	Deep, sandy to gravelly soils on flat to moderately sloping terrain with open vegetative cover	Best 1995; MGSWG 2011

Mohave ground squirrel prefers deep, sandy to gravelly soils on flat to moderately sloping terrain and will avoid rocky areas for the most part (Best 1995; MGSWG 2011). The species is not known to occupy areas of desert pavement (MGSWG 2011). Soil characteristics are particularly important because Mohave ground squirrels construct burrows to provide temperature regulation, avoid predators, and use during the inactive season (75 FR 22063–22070).

Foraging Requirements

The Mohave ground squirrel primarily feeds on plant material. In the short term, they specialize in foraging on certain plant species, but as these sources become less available throughout the active season, the Mohave ground squirrel adapts its foraging strategy to maximize energy intake, exploiting food sources that are intermittently available (75 FR 22063–22070). High water content may be a component of their food selection as plants are eaten at different times depending on their water content (Best 1995; 75 FR 22063–22070). Mohave ground squirrels consume the leaves, fruits, and seeds of a variety of annual and perennial plants, fungi, arthropods, including butterfly larvae. At various times of the year and depending on location, they may consume leaves, forbs, shrubs, and grasses of several species and genera, including creosote (*Larrea tridentata*), winter fat (*Krascheninnikovia lanata*), spiny hop-sage (*Grayia spinosa*), freckled milk-vetch (*Astragalus lentiginosus*), eremalche (*Eremalche exilis*), desert-marigold (*Baileya pleniradiata*),

langloisia (*Langloisia setosissima*), Mojave monardella (*Monardella exilis*), saltbush (*Atriplex* spp.), gilia (*Gilia* spp.), golden linanthus (*Linanthus aureus*), and Mediterranean grass (*Schismus arabicus*), as well as seeds of box thorn (*Lycium* spp.) (Best 1995; 75 FR 22063–22070; MGSWG 2011). On the Coso Range (outside of the Plan Area), about 42% of the species' diet, based on fecal samples, consisted of forbs and shrub material (primarily foliage) (MGSWG 2011). Shrubs are especially important both early and late in the active season when forbs are not available (MGSWG 2011). Winter fat, spiny hop-sage, and saltbush made up 60% of the species' shrub diet, indicating that these species are the main food source when forbs are unavailable (MGSWG 2011). It has been suggested that habitats where winter fat and hop-sage are absent may be suboptimal for Mohave ground squirrel (MGSWG 2011).

Reproduction

The Mohave ground squirrel breeding season is from mid-February to mid-March (Best 1995; Laabs 2006) (Table 3). Males emerge from hibernation in February, up to two weeks before females, and during this time they may be territorial (Best 1995). Females generally only occupy male territories for one or two days then establish their own home ranges after copulation. Recent radiotelemetry data indicate that males expand their activity areas the breeding to overlap several established female ranges, (unpublished data, Leitner, pers. comm. 2012). Males stake out the overwintering sites of females to mate with them when they emerge (MGSWG 2011).

Pregnant females are present from March through April (Leitner, pers. comm. 2012) and gestation lasts from 29 to 30 days (Best 1995). Litter sizes range from four to nine (Best 1995), though mortality of juveniles is high during the first year, especially for juvenile males (MGSWG 2011). Parental care and lactation continues through mid-May. Litters generally appear above ground in early May (Harris and Leitner 2004). Females will breed at 1 year of age if environmental conditions are suitable, but males do not mate until 2 years of age (MGSWG 2011).

The amount of fall and winter precipitation generally determines Mohave ground squirrel reproductive success. In low rainfall years (e.g., less than 6.5 cm [2.6 in.]), they may forego breeding (MGSWG 2011), and breeding may not occur for several years during prolonged drought (Best 1995). Because of the small geographic range of the

species, low rainfall can lead to reproductive failure throughout the range (MGSWG 2011). During these periods, all available forage may be converted to body fat and squirrels can enter dormancy as early as April (Leitner 1999).

Table 3. Key Seasonal Periods for Mohave Ground Squirrel

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Aestivation								X	X	X		
Hibernation	X										X	X
Breeding		X	X									
Parental Care			X	X	X							

Notes: Aestivation is the summer period of inactivity and hibernation is the winter period of inactivity.

Sources: Best 1995; Laabs 2006.

Spatial Behavior

The Mohave ground squirrel is generally only active above ground between February and July (MGSWG 2011), but the active period may begin as early as mid-January (Harris and Leitner 2004). Adults generally enter aestivation earlier than juveniles (MGSWG 2011). Timing of emergence varies geographically as it appears to depend on temperature and elevation (Gustafson 1993; Laabs 2006). Furthermore, the timing of emergence and length of the active season varies by sex, age, and availability of food resources (MGSWG 2011). Adult females and juveniles generally have longer active seasons than adult males. The active season is also longer when there is more food available, which is often correlated with greater precipitation (MGSWG 2011). Mohave ground squirrels are diurnal, spending much of the day above ground during the active season. During the inactive season, Mohave ground squirrels remain underground in burrows and enter a state of torpor (a state of reduced physiological activity or sluggishness) to conserve their energy reserves and water (Best 1995; MGSWG 2011).

Harris and Leitner (2004) conducted a 5-year radiotelemetry study of home range use by Mohave ground squirrels in the Coso Range in Inyo

County. At this study site, individual Mohave ground squirrel home ranges (calculated using both minimum convex polygon and adaptive kernel methods) varied substantially by year, individual, sex, and season (i.e., mating season vs. post-mating season) (Table 4). Generally, males have larger home ranges than females, with the most pronounced differences during the mating season. Female ranges expanded during the postmating season compared to the mating season (Table 4). In drought years when reproduction did not occur, female postmating season home ranges varied inversely in relation to precipitation, which in turn is related to the amount of available forage (Harris and Leitner 2004). Female home ranges contracted in years of moderate drought and lack of reproduction, which may be a strategy to reduce energy expenditure and enter dormancy sooner (Harris and Leitner 2004). During years of high precipitation and successful reproduction, female postmating home ranges were larger in response to the need for more energy sources to support gestation and lactation (Harris and Leitner 2004). Females that were radio tracked for more than 1 year showed a high level of home range site fidelity and all individuals' home ranges exhibited overlap over different years; i.e., no females moved to entirely new home ranges (Harris and Leitner 2004).

Table 4. Mohave Ground Squirrel Home Ranges in the Coso Range¹

Type	Median MCP Home Range ²	Citation
Mating Season Home Range – Male	16.63 acres (range: 10.5–99.1 acres)	Harris and Leitner 2004
Mating Season Home Range – Female	1.83 acres (range: 0.70–2.3 acres)	Harris and Leitner 2004
Postmating Home Range – Male	3.06 acres ³	FR 22063–22070
Postmating Home Range – Female	2.96 acres ³	FR 22063–22070

Notes:

¹ The Coso Range is located north of the Plan Area

² MCP = minimum convex polygon

³ The home range statistics reported in FR 22063–22070 (the 90-day finding on the petition to list the species) cite Harris and Leitner (2004), but the original paper does not appear to include these specific statistics for postmating home ranges. While these statistics appear to be consistent with Figure 1 in Harris and Leitner (2004) and are consistent with the text description of postmating home ranges, they cannot be confirmed by a review of the original paper and it is unclear how these statistics were generated for the 90-day finding on the petition.

Male home ranges during the mating season were very large and reflected long-distance movements large enough to cross the home ranges of several females (Harris and Leitner 2004). Long-distance movements (> 656 feet) were much more frequent during the mating season compared to the postmating season, and females seldom made such long movements (Harris and Leitner 2004).

Mohave ground squirrels maintain three types of burrows within their home ranges: (1) home burrows that are used overnight during the active season and usually located at the edge of a home range; (2) aestivation burrows; and (3) accessory burrows that are used during social interactions or for escape and thermoregulation during the midday (Best 1995). Burrows are typically constructed under large shrubs (MGSWG 2011).

Harris and Leitner (2005) used radiotelemetry to track dispersal movements by juvenile Mohave ground squirrels in their first year to hibernation sites. Most juveniles dispersed relatively long distances from their natal burrow area, and exhibited dispersal that is farther than other squirrels and other mammals in proportion to home range sizes (Harris and Leitner 2005). Mean male dispersal from the natal area was 9,580 feet (range: 0 to 20,439 feet) and mean female dispersal from the natal area was 2,470 feet (range: 0 to 12,670 feet) (Harris and Leitner 2005). However, with the exception of the one female that moved 12,760 feet to a hibernation site, all the females dispersed less than 1,640 feet from the natal area, indicating that juvenile dispersal is male-biased (Harris and Leitner 2005). Notably, the juveniles that dispersed more than 2,160 feet moved out of the alluvial basin where the study was located and had to cross rocky terrain with low shrub cover, which is not considered suitable habitat for the species, and at least two individuals crossed dirt roads (Harris and Leitner 2005). In addition, all but one of the individuals dispersing more than 2,160 feet left the natal area on a particular day and did not return to the natal area (Harris and Leitner 2005). Shorter dispersal movements may involve exploratory movements where juveniles return to the natal area at night before a permanent move. Harris and Leitner (2005) suggest that the relatively mobile behavior of juvenile Mohave ground squirrels may have adaptive value for connecting location populations and recolonizing sites that have experienced natural local extinctions (e.g., due to prolonged drought).

Ecological Relationships

There is little direct information on the potential role of Mohave ground squirrels in maintaining ecological relationships and processes. Their burrow systems likely provide refuge for other species that do not dig their own burrows such as snakes and lizards and potentially other small rodents. The range of the Mohave ground squirrel is entirely overlapped by the diurnal white-tailed antelope squirrel (*Ammospermophilus leucurus*), but there appears to be little direct competition between the two species (MGSWG 2011). While Mohave ground squirrels primarily forage on the foliage of shrubs and forbs, and secondarily on the seeds of shrubs and forbs, the antelope squirrel exhibits the opposite behavior of concentrating on seeds of forbs and shrubs and insects (about 25% of their diet) and secondarily foraging on foliage (MGSWG 2011). The Mohave ground squirrel is behaviorally dominant over the antelope squirrel (MGSWG 2011). As primarily a seed-eater, the antelope squirrel is also active on the surface year round (MGSWG 2011). Potential competitive relationships with birds, herbivorous reptiles (e.g., desert tortoise), or ants for food resources are unknown. They are probably prey for several natural predators, such as coyote (*Canis latrans*), American badger (*Taxidea taxus*), bobcat (*Lynx rufus*), red-tailed hawk (*Buteo jamaicensis*), golden eagle (*Aquila chrysaetos*), prairie falcon (*Falco mexicanus*), common raven (*Corvus corax*), and Mojave rattlesnake (*Crotalus scutulatus*) (Best 1995).

Population Status and Trends

Global: Moderate decline to relatively stable (NatureServe 2011)

State: Same as above

Within Plan Area: Same as above

Data are lacking to assess population abundance and trends for the Mohave ground squirrel (76 FR 62219). Systematic or sample-based surveys in the species' range have not been conducted at a level that allow for population estimates and comparisons over time. As discussed in Distribution, the species likely has been extirpated from portions of its former range due to urban and agricultural development, especially around the Lancaster, Palmdale, and Victorville areas.

Threats and Environmental Stressors

The primary threat to the Mohave ground squirrel has been habitat loss and fragmentation (Leitner 2008; MGSWG 2011). The Mohave ground squirrel's range has been reduced or its habitat destroyed and degraded by urban and rural development on private and public lands, agricultural development, military activities, energy projects, and transportation (Leitner 2008; MGSWG 2011; 76 FR 62214–62258). For energy projects, large-scale solar projects are particularly destructive to Mohave ground squirrel habitat because they have a large disturbance footprint and they are sited on level and gently sloping terrain that is characteristic of Mohave ground squirrel habitat (76 FR 62214–62258).

Livestock grazing and off-highway vehicles (OHVs) may also cause habitat degradation and have direct impacts on Mohave ground squirrel (Leitner 2008; MGSWG 2011; 76 FR 62214–62258).

Grazing by cattle and sheep can affect vegetative structure, disturb soils, accelerate erosion, and collapse burrows (MGSWG 2011). Cattle and sheep forage on winter fat foliage, which is also important to Mohave ground squirrel, especially in years with low precipitation and annual forb production (MGSWG 2011). Although livestock grazing is listed as a potential threat to Mohave ground squirrel, the BLM has been eliminating or reducing grazing in some areas of the species range (76 FR 62237) and grazing does not occur on military lands, state parks or CDFW ecological reserves (Leitner, pers. comm. 2012). The USFWS 12-month finding on October 6, 2011 conclude that livestock grazing is not currently a threat to the Mohave ground squirrel (76 FR 62214–62258).

OHV use is a threat to Mohave ground squirrel through direct collisions, disturbance of soil, destruction of shrubs, and facilitation of invasive species that displace native species along dirt roads and trails (MGSWG 2011). The West Mojave Plan Route Designation report indicates that 47% of 310 vegetation transects are bisected by some type of off-road vehicle track (MGSWG 2011). The four BLM-operated off-highway areas (Jawbone Canyon, Dove Springs, El Mirage, and Spangler Hills) cover over 417 km² (161 mi²) within the Mohave ground squirrel's range (MGSWG 2011).

Prolonged drought is another threat to the Mohave ground squirrel. Low rainfall causes reduced productivity of annual plants, which can cause Mohave ground squirrels to forego breeding during drought periods because insufficient energy is available to support gestation and lactation (Best 1995; Harris and Leitner 2004). Local population extinction can result with prolonged drought events that suppress reproduction for several years (Best 1995). Prolonged drought events alone would not pose a serious threat to the species, considering its likely adaptations for these conditions, such as prolonged aestivation and long dispersal movements that allow for recolonization (Best 1995; Harris and Leitner 2005). However, habitat loss, fragmentation, and degradation can preclude recolonization of habitat from which local populations have been extirpated as a result of drought because the sites become functionally isolated from occupied areas (Laabs 2006).

Urban and rural uses have introduced potential impacts to Mohave ground squirrel that may occur where habitat is near development. Domestic cats (*Felis catus*) and dogs (*Canis familiaris*) may be predators and the use of rodenticides and pesticides around agricultural fields, golf courses, earthen dams, and canal levees may directly affect the species (MGSWG 2011).

Although common raven is a natural predator, their populations have increased substantially within the Mohave ground squirrel's range and they are a known predator for small mammals (MGSWG 2011). Therefore, ravens may be exerting higher predation pressure on the species than occurred historically.

Conservation and Management Activities

Conservation and management planning for the Mohave ground squirrel has been ongoing on several fronts, including by the West Mojave Plan; CDFW; the Desert Managers MGSWG; and on military installations.

The West Mojave Plan establishes a 1,726,712-acre (2,698 mi²) Mohave ground squirrel Conservation Area on non-military public and private lands for the long-term survival and protection of the species. The Conservation Area covers about 41% of the estimated current range of the species. Public lands within the Conservation Area would be designated as a BLM Wildlife Habitat Management Area. The West Mojave Plan established two goals for Mohave ground

squirrel: Goal 1, ensure long-term protection of Mohave ground squirrel habitat throughout the species' range; and Goal 2, ensure long-term viability of the species throughout its range. The West Mojave Plan also established several objectives to meet these goals.

For Goal 1, the West Mojave Plan objectives are:

- Establish a Conservation Area for the protection of unfragmented habitat outside military installations (noted previously)
- Establish biological transition areas to minimize indirect impacts of human development on the Conservation Area
- Allow for adjustment of the Conservation Area boundary based on scientific studies
- Implement actions to ensure long-term protection of habitat for Mohave ground squirrel in the Conservation Area throughout the life of the Plan
- Annually track the loss of Mohave ground squirrel habitat resulting from Plan implementation
- Cooperate with military installations in sharing scientific information and reviewing management plans to assist managers in evaluating Mohave ground squirrel habitat protection on the installations.

For Goal 2, the West Mojave Plan objectives are:

- Per CDFW mandate, minimize and fully mitigate the impacts of the Plan's incidental take of Mohave ground squirrel throughout the life of the Plan
- Upon Plan adoption, implement studies that would determine four measureable biological parameters for the Mohave ground squirrel: (1) regional status; (2) potential "hot spots" (refugia); (3) genetic variation throughout the species' range; and (4) the species' ecological requirements
- Establish long-term study plots throughout the species' range to annually monitor populations, and fund continued monitoring in the Coso Range (outside of the Plan Area) to provide baseline information

- Use the biological information from the above objectives to modify management prescriptions, as warranted, to ensure the long-term viability of the species.

To date, CDFW has spent approximately \$800,000 funding studies that include information on genetics, diet, dispersal, and location of Mohave ground squirrels over the past several years. Also, approximately \$100,000 from Section 2081 incidental permits has or will fund Mohave ground squirrel trapping administered by the Desert Tortoise Preserve Committee (MGSWG 2011).

The military has also conducted activities to inform conservation and management of the Mohave ground squirrel.

Edwards Air Force Base has completed at least 3 years of Mohave ground squirrel inventories and has monitored 60 Habitat Quality Analysis plots. Since 2003, approximately 45% of the Edwards Air Force Base has been surveyed and funds are programmed for Mohave ground squirrel inventories through 2013 (MGSWG 2011).

The National Training Center (NTC) and Fort Irwin contain 445,241 acres of Mohave ground squirrel habitat. The NTC and Fort Irwin funded trapping studies for the Mohave ground squirrel in 1977, 1985, and from 1993 to 1994. The MGSWS (2011) suggests that the three conservation areas for Lane Mountain milk-vetch (*Astragalus jaegerianus*) on Fort Irwin will work well for Mohave ground squirrel conservation. In addition, under an agreement with CDFW, the Paradise Conservation Area will be enhanced for Mohave ground squirrel by planting the species' preferred food plants (MGSWG 2011). However, at present there is no evidence that these areas support the Mohave ground and, further, these areas are generally rocky and hilly with little of the alluvial soils needed by the species (Leitner, pers. comm. 2012). There is currently no evidence that food enhancement is successful in the Paradise Conservation Area (Leitner, pers. comm. 2012).

Data Characterization

Because Mohave ground squirrel is inactive much of the year, and squirrel abundance and the length of the active season varies from year to year (MGSWG 2011), even when studies are scheduled carefully they may not be able to establish the presence or absence of the species from a site with a high level of certainty. Further, if unfavorable conditions (little fall and winter precipitation) persist for several seasons, local extirpation can occur, but re-colonization of these areas under more favorable conditions can occur. In addition, the species is not distributed continuously throughout its range independent of proposed habitat conversion (MGSWG 2011). Because trapping studies typically are sited in habitat proposed for conversion, grids and transects are not randomly or systematically placed in a manner that samples across the range of potentially suitable habitats and allows for inferences about occupation throughout the species' range. Many of the trapping studies for Mohave ground squirrel have been concentrated south of State Route 58 where most of the habitat conversion has been proposed (Leitner 2008). For this reason, there are extensive areas of the Mohave ground squirrel's range in the Plan Area that have not been studied and the species' status is unknown (Leitner 2008).

Management and Monitoring Considerations

Protection of large core areas of native habitat and adequate connections among the core areas are required to ensure the long-term survival and recovery of the Mohave ground squirrel. Ideally, biological, demographic, and genetic considerations should govern the size and location of preserve areas. As an initial recommendation for habitat conservation of currently occupied habitat, Leitner (2008) defines core areas for the species based on three objective and measureable criteria:

1. Demonstrated species persistence in an area over a long time period on the order of two to three decades;
2. Species must be currently present in multiple locations within the core area; and
3. There are substantial numbers of adults forming a viable reproductive population.

With these criteria in mind, core preserve areas need to be large enough to support populations that are resilient to natural fluctuations in size that occur in relation to precipitation patterns, including prolonged drought. Each population has to be large enough to withstand several years of no or reduced reproduction; if a drought extends so long that no reproduction occurs over a 4- or 5-year period, even the youngest cohort would likely die of old age before reproducing. Therefore, large preserve areas are needed to minimize the risk of local extinction from demographic and environmental stochastic events, as well as from the genetic problems associated with small population size, such as loss of genetic variability, genetic drift, and inbreeding depression. Smaller areas are also more susceptible to edge effects and disturbance from surrounding non-compatible land use (Laabs 2006).

Core reserves in high-quality habitats are required to support populations of the species during drought conditions and that can provide sources from which populations may expand when conditions are favorable to the species. Research conducted on the Coso Range (outside of the Plan Area) found that certain shrub species (winter fat and spiny hop-sage) appear to be important in providing forage when annual forb growth is low and thus may be critical to the persistence of populations during drought years (MGSWG 2011). However, these data are primarily from a study site at the north edge of the species' range and community (Mojave Mixed Woody Scrub) that is somewhat atypical of the majority of the species' range. Additional research into food habits and critical habitat features in creosote bush scrub and saltbush scrub habitats is needed to identify critical habitat features (Laabs 2006).

Based on the three objective criteria cited previously, Leitner (2008) identified four core areas, as summarized in Table 5. It is important to note that these core areas are only those identified so far and that with more survey data other areas may meet the objective criteria for a core area (Leitner 2008).

Table 5. Mohave Ground Squirrel Core Areas

Core Area Name	Area (acres)	Number of Positive Records (1998–2007)
Coso/Olancha	111,690	33
Little Dixie Wash	97,112	44
Coolgardie Mesa/Superior Valley	127,450	23
Edwards Air Force Base	76,761	34

Source: Leitner 2008.

As a rare species with apparent disjunct local populations, preserving naturally occurring genetic variability is critical to the preservation of the Mohave ground squirrel. Connectivity between preserve areas will be important to maintain gene flow between local populations and facilitate recolonization of areas if local extinctions occur. According to Leitner (2008), the four core areas identified are isolated from each other by distances that range from 30 to 50 miles. Leitner (2008) identified conceptual linkages between the corridors. Demographic considerations, such as home range size and average dispersal distances, should determine the width of connectivity corridors (Laabs 2006). As described previously, Mohave ground squirrels are capable of dispersing relatively long distances; the maximum juvenile male dispersal was about 3.9 miles and the maximum female dispersal was about 2.4 miles (Harris and Leitner 2005). With distances between core habitat areas of 30 to 50 miles (Leitner 2008), substantial swaths of suitable habitat between core areas will therefore be necessary.

The habitat management component of the *Draft Mohave Ground Squirrel Conservation Strategy* (MGSWG 2011) focuses on limiting habitat loss through effective conservation measures, mitigation, and compensation by avoiding and minimizing impacts to Mohave ground squirrel and its habitat and restoring and enhancing habitat. The strategy also focuses on securing and managing sufficient core habitat and corridors to maintain self-sustaining populations (MGSWG 2011). The West Mojave Plan also focuses on establishing conservation areas to protect unfragmented habitat and biological transition areas to protect conservation areas from indirect human impacts. The West

Mojave Plan includes objectives for implementing biological studies regarding the species' range, hot spots, and ecological requirements. This information would be used to inform conservation and management of the species.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mohave ground squirrel, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 3,501,554 acres of modeled suitable habitat for Mohave ground squirrel in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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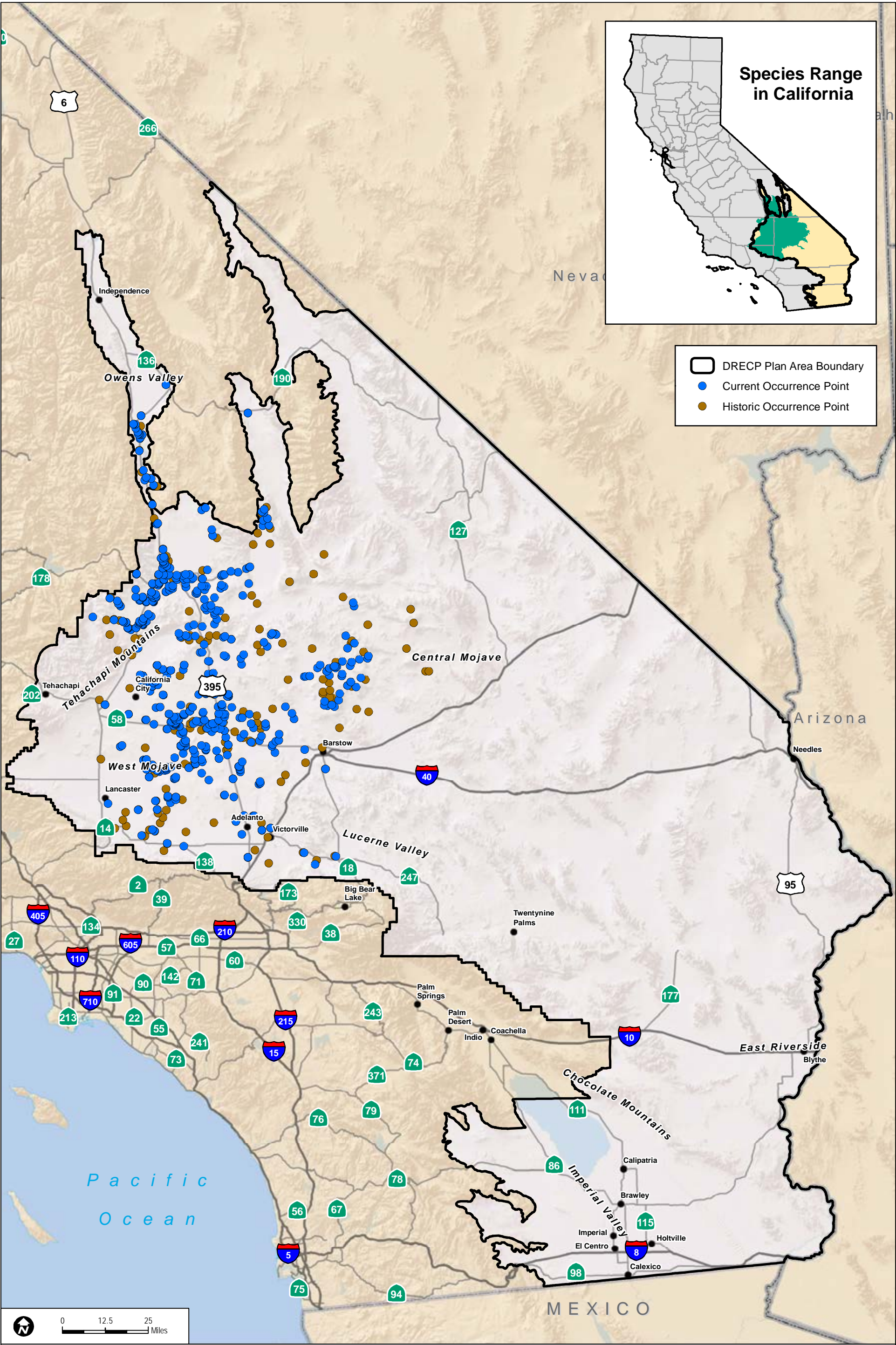
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M05

Mohave Ground Squirrel Occurrences in the Plan Area

Desert Renewable Energy Conservation Plan (DRECP) Baseline Biology Report

October 2015

Pallid Bat (*Antrozous pallidus*)

Legal Status

State: Species of Special Concern

Federal: Bureau of Land
Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A

Notes: None



Photo courtesy of Scott Trageser.

Taxonomy

The pallid bat (*Antrozous pallidus*) is the only species in the genus *Antrozous* of the family Vespertilionidae (Hermanson and O'Shea 1983; Hoofer et al. 2003) (*Antrozous* formerly included *A. dubiaquercus*, but this Central American species is now assigned to the genus *Bauerus* [Hermanson and O'Shea 1983]). A study of phylogenetic relationships of plecotine bats using mitochondrial ribosomal sequences supported the placement of pallid bat as a single-species genus in the family Vespertilionidae (Hoofer et al. 2003). There are seven recognized subspecies of pallid bat (Wilson and Reeder 2005), of which *A. p. pallidus* is likely the subspecies present in the Desert Renewable Energy Conservation Plan (DRECP) project Plan Area, although *A. p. pacificus* may also occur in the western portion of the Plan Area (Hall 1981). The status of pallid bat as California Species of Special Concern is for the full species *A. pallidus*, so a subspecific assignment is not relevant to the conservation of this species in the Plan Area. No other available information indicates other important taxonomic considerations. The species' physical characteristics are described in detail in Hermanson and O'Shea (1983).

Distribution

General

The pallid bat is widespread throughout the western United States; southern British Columbia, Canada; and mainland and Baja California, Mexico (Hermanson and O'Shea 1983; Hall 1981). Within the United States, it ranges east into southern Nebraska, western Oklahoma, and western Texas (Figure SP-M06). The pallid bat is locally common in the Great Basin, Mojave, and Sonoran deserts (especially the Sonoran life zone) and grasslands throughout the western United States, and it also occurs in shrublands, woodlands, and forests at elevations up to 2,440 meters (8,000 feet) (Hermanson and O'Shea 1983; Hall 1981). The pallid bat occurs throughout California, except at the highest elevations of the Sierra Nevada range. Although this species prefers rocky outcrops, cliffs, and crevices with access to open habitats for foraging, it has been observed far from such areas (Hermanson and O'Shea 1983).

Distribution and Occurrences within the Plan Area

Historical

The DRECP database for pallid bat, composed of Bureau of Land Management (BLM) and California Natural Diversity Database (CNDDB) (CDFW 2013) records, and observations by Brown (CDFW 2013; Dudek 2013), includes 20 historical records (i.e., pre-1990) for the Plan Area, dating from 1911 to 1981, and two with an unknown observation date. An additional 11 records are from areas within 5 miles of the Plan Area boundary. The historical occurrences in the Plan Area include the southern Owens Valley–eastern Sierra Nevada–Inyo Mountains area, the Mesquite Mountains in eastern San Bernardino County, the Twentynine Palms area, the lower Colorado River, and the Salton Sea area.

See Figure SP-M06 for historical and recent occurrences of pallid bat in the Plan Area.

Recent

There are 40 recent (i.e., since 1990) records in the Plan Area and 10 additional records within the 5-mile buffer area around the Plan Area (CDFW 2013; Dudek 2013). The geographic areas of recent occurrences are similar to the historical occurrences, with small clusters of observation in the Owens Valley–eastern Sierra Nevada area, Providence Mountains, Kingston Range, Avawatz Mountains, Cady Mountains, Twentynine Palms area, Little San Bernardino Mountains, Hexie Mountains, the Lower Colorado River, Chocolate Mountains, and the Peninsular Range in east San Diego County.

As with the historical data, the specificity of these recent occurrence data is variable, with some records identifying roosts and others only including general location information for observations. This dataset, therefore, should be viewed as reflecting the recent documented distribution of the species in the Plan Area and should not be used as detailed data for specific roost sites.

Natural History

Habitat Requirements

Pallid bat day roosting habitat typically includes rocky outcrops, cliffs, and spacious crevices with access to open habitats for foraging (Hermanson and O'Shea 1983; Vaughan and O'Shea 1976). Pallid bats may also roost in caves, mines, bridges, barns, porches, and bat boxes, and even on the ground under burlap sacks, stone piles, rags, baseboards, and rocks (Beck and Rudd 1960; Rambaldini 2006). Radiotelemetry data has also shown that in the desert pallid bats will roost in holes on the ground and in rock crevices on creosote bush flats, not just in mountain ranges (Brown, pers. comm. 2012). Up to the late 1940s, they were common in buildings at low elevations of the South Coast Ecoregion (Miner and Stokes 2005). For example, in the Newhall area of Southern California, they recently were observed using buildings for both day and night roosts (Johnson 2006). In Northern California, they were observed using buildings and large-diameter, tall, live trees and snags in mature forest stands for both day and night roosting (Baker et al. 2008). In Baker et al. (2008), live trees and snags used for roosting were consistently tall in height, large in diameter, and located in mature stands in micro-sites with

low percentages of overstory and mid-story cover. Day roosts generally are warm, have obstructed entrances and exits, and are high enough to avoid terrestrial predators (Rambaldini 2006). A study of night roosts, including rock overhangs, bridges, and buildings, in Oregon found that they were protected from rain and allowed free flight space for bats in and out of the roost (Lewis 1994).

Although pallid bats may use a variety of roosting habitats, they are also selective of roost sites with microenvironments that minimize energy expenditure through adaptive hypothermia and maintain low metabolic rates (Vaughan and O'Shea 1976). In spring and fall at roost sites in Central Arizona, they used vertical crevices that passively warmed during the afternoon prior to emergence, and in the summer, they used deep horizontal crevices that acted as heat sinks and kept ambient temperatures low (Vaughan and O'Shea 1976). A roost temperature of about 30 degrees Celsius (86 degrees Fahrenheit) is considered about optimal for maintaining low metabolic rates (Trune and Slobodchikof 1976; Vaughan and O'Shea 1976). In desert regions, roost sites are often near water, although they have been observed in areas without apparent water sources (Hermanson and O'Shea 1983).

Pallid bat day roosts consisting of single- or mixed-sex colonies usually are established in crevices or man-made structures. Day roosts usually have at least 20 individuals and sometimes more than 200 individuals (Hermanson and O'Shea 1983).

Foraging habitats for pallid bats are varied and include grasslands, oak savannah woodlands, open pine forests, talus slopes, and agricultural areas (Rambaldini 2006). In a study of bat use of riparian habitats in southern Nevada, including riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland, Williams et al. (2006) recorded about 88% of pallid bat occurrences in riparian woodland. Although most foraging probably occurs in close proximity to night roosts, movements greater the 2 kilometers (1.2 miles) from roosting sites in forest habitats are common (Baker et al. 2008), and movements up to 30 kilometers (18.6 miles) have been recorded (Hermanson and O'Shea 1983). See discussion in Spatial Behavior for more information.

Table 1 summarizes the likely habitat associations for pallid bat in the Plan Area.

Table 1. Habitat Associations for Pallid Bat

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Rocky, Barren, and Unvegetated Community	Day and night roosts	Day and night roosting	>50% rocky slopes within 6.2 miles of water source	Hermanson and O'Shea 1983
All natural land covers (i.e., except developed and disturbed)	Foraging	Primary foraging	Natural land covers within 3.1 miles of day roosting habitat	Baker et al. 2008; Bell 1982; Rambaldini 2006
All natural land covers (i.e., except developed and disturbed)	Foraging	Secondary foraging	Natural land Covers 3.1 to 6.2 miles of day roosting habitat	Baker et al. 2008; Bell 1982; Rambaldini 2006

Notes: Water sources include major rivers, reservoirs, lakes, ponds, seeps and springs, and perennial streams. Pallid bats are expected to forage in virtually all relatively open, natural land covers in the Plan Area where suitable prey are present.

Foraging Requirements

Pallid bats forage about 0.5 to 2.5 meters [1.6 to 8.2 feet] above the ground surface, and their foraging behavior is directed toward prey that are close to the ground, on the ground, or perched on exposed vegetation (O'Shea and Vaughan 1977). They may forage both aerially and by gleaning from plants, and they have also been observed to take prey by crawling along the ground. Their diet generally has been described to include scorpions, ground crickets, solpugids, darkling ground beetles, carrion beetles, short-horned grasshoppers, cicadas, praying mantids, long-horned beetles, and sphingid moths (Hermanson and O'Shea 1983). While pallid bats are primarily insectivores, they have also been observed to eat lizards and smaller

bats in captivity (Hermanson and O'Shea 1983) and likely take a variety of small vertebrates in the wild. Their specific diets vary geographically and may reflect genotypic or phenotypic selection (Johnston and Fenton 2001). Pallid bats generally take large prey (up to 6.0 centimeters [2.4 inches] total body length) (O'Shea and Vaughan 1977). In both a coastal area (Marin County) and a desert area (Caliente Mine in Death Valley) in California, pallid bats foraged for Orthoptera (grasshoppers, crickets) and Coleoptera (beetles), and smaller percentages of Solpugida (sun scorpions), Lepidoptera (moths), and Diptera (flies). At Caliente Mine, Coleoptera made up about 55% of their diet by volume, but diet changed over time, reflecting the availability of prey. Individuals in the local population tended to have the same diet at any given time (Johnston and Fenton 2001). In contrast, at the Marin County site, diets were varied, but the variation was related to individual differences (i.e., there was no "average" diet for the group such as that of the Caliente site), and these differences may have reflected learning that reduces searching and handling time (Johnston and Fenton 2001).

Reproduction

Pallid bats breed in October through December, and possibly through February (Hermanson and O'Shea 1983) (Table 2). Females store sperm and ovulation occurs during the following spring. Gestation is approximately 9 weeks, and birth in the southwestern United States typically occurs from May through June (Hermanson and O'Shea 1983). Litter size is typically 2 young (approximately 80% of litters (Bassett 1984)), and occasionally 3; yearling females may breed but litter size is 1 (Davis 1969; Hermanson and O'Shea 1983). The young are born relatively undeveloped, but they mature rapidly and engage in their first flight at 33 to 36 days (Davis 1969). They achieve full adult flight capability by about 49 days of age and full adult weight by 56 days of age (Hermanson and O'Shea 1983). Yearling males are not sexually active their first autumn and probably not their first year (Davis 1969). Mature males and females have the same body dimensions (e.g., weight, forearm length, wing area); they do not exhibit sexual dimorphism (Davis 1969).

Pallid bats have lived up to 9 years in captivity (Hermanson and O'Shea 1983).

Table 2. Key Seasonal Periods for Pallid Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	?	?								x	x	x
Birth/ Development					x	x	x	x				
Winter Torpor	x	x	x	x								x

Sources: Bassett 1984; Davis 1969; Hermanson and O'Shea 1983.

Spatial Behavior

Pallid bats in central Arizona exhibited a bimodal foraging activity pattern, with two foraging bouts separated by a period of night roosting, with the timing and duration of these activities seasonally variable (O'Shea and Vaughan 1977). During the summer months, time away from the roost varies between approximately 45% to 58% of the night. In September and October, time away from the roost varies between 25% to 27% of the night. (O'Shea and Vaughan 1977) Pallid bats may be active outside the roost any time of year, but their activity during the winter may be erratic, which probably is associated with cold periods when they are in torpor (Table 2). They have been mist-netted at temperatures as low as 2 degrees Centigrade (35.6 degrees Fahrenheit) in southern Nevada (O'Farrell et al. 1967). In contrast to O'Shea and Vaughan (1977), O'Farrell et al. (1967) did not detect a bimodal activity period in southern Nevada during the fall and winter; all captures were 1.5 to 5 hours after dusk. This information indicates that nightly foraging activity by pallid bats is seasonally variable.

During July through August, pallid bats in central Arizona showed little fidelity to specific roosting sites, but during the cooler months they showed greater fidelity to certain roosting sites (O'Shea and Vaughan 1977), which may reflect more specific roost requirements during the colder months to maintain thermoregulation (also see Habitat Requirements regarding day roost characteristics).

The distances that pallid bats travel during foraging bouts may be limited by the availability of night roosts because they frequently bring large prey to these sites where it is then eaten (O'Shea and Vaughan 1977). Bell (1982), for example, observed pallid bats foraging within 3 kilometers (1.9 miles) of roost sites in desert grasslands in New Mexico. A radio-tracking study in British Columbia found that foraging occurred within 1.5 kilometers (0.9 mile) of day roost sites (Rambaldini 2006). In this study, males returned to the day roost for short periods between foraging bouts (Rambaldini 2006) (however, note from discussion above that nightly foraging activity is seasonally variable). In coniferous forest in Northern California, radio-tracking documented that foraging bouts more than 2 kilometers (1.2 miles) from the day roost were common, but most foraging occurred in close proximity to day roosts (Baker et al. 2008). The longest distance moved during this study was 4.7 kilometers (2.9 miles) by a pregnant female. Lactating females had average foraging ranges of 1.56 square kilometers (0.6 square mile), and post-lactating females had average ranges of 5.97 square kilometers (2.3 square miles) (Baker et al. 2008). However, flights up to 30 kilometers (19 miles) between night roosts have been recorded, indicating that pallid bats have the capacity to fly long distances. Further, homing studies have shown a maximum return distance of 174 kilometers (108 miles), and several recoveries have shown return distances of 48 to 51 kilometers (30 to 32 miles) from release sites within 7 to 8 hours after release (Hermanson and O'Shea 1983).

Dispersal flights in the central Arizona study occurred in mid-August and were characterized by straight-line flight movements from the day roost (in contrast to the typical circling of the roost area) at approximately 25 meters (82 feet) above the ground and no evidence of foraging (O'Shea and Vaughan 1977). These dispersal flights occurred at the same time the population numbers at the day roost sharply declined (O'Shea and Vaughan 1977), indicating that young were leaving the maternity site.

Ecological Relationships

Day roost selection, fidelity, and lability (flexibility) by pallid bats indicate potentially important ecological relationships and are region-specific. As discussed in Habitat Requirements, pallid bats select day

roosts that appear to maximize adaptive hypothermia (Vaughan and O'Shea 1976). In addition to microclimate stability, deep crevices used for day roosts may provide protection from predators and protection of juveniles that may fall from the ceiling (Lewis 1995). In central Arizona, where such deep crevices are available, females change day roosts in the spring, but not during pregnancy and lactation (O'Shea and Vaughan 1977). In Oregon, where such deep crevices are not available for roosting, females change day roosts throughout the summer (Lewis 1995). Lewis (1995) suggests that the Oregon populations benefit from roost lability by reducing ectoparasite infestations. In Arizona, the benefits of roost fidelity to the deep crevices may outweigh the impacts of ectoparasites (Lewis 1995).

In addition to selecting roosting sites to maximize adaptive hypothermia, social roosting also appears to be important for conserving metabolism. An experimental study showed that individual roosting bats had higher metabolic rates and weight loss than bats roosting in clusters and at suboptimal temperatures of 25 and 35 degrees Celsius (77 and 95 degrees Fahrenheit) (Trune and Slobodchikoff 1976).

Pallid bats may share both day and night roosts with other bat species such as Brazilian free-tailed bat (*Tadarida brasiliensis*) and Yuma myotis (*Myotis yumanensis*) (Hermanson and O'Shea 1983; Licht and Leitner 1967), but there is no evidence in the literature of competitive or symbiotic relationships with other bats. Congregations with other bat species at both day and night roosts may simply reflect use of limited resources.

Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce inter-specific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-, within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering.

Compared to other bat species, pallid bats emerge from day roosts relatively late in the evening (Hermanson and O'Shea 1983), but there is no information to suggest that this reflects competition for prey with other species. Artificial lighting may affect competitive predator-

prey relationships among bats. Longcore and Rich (2004) suggest that artificial lighting, which attracts many insects taken by bats, including moths (Frank 1988), may alter local community relationships because the faster-flying bats congregate around lights and can exploit this concentrated food source while slower-flying bats avoid lights and are unable to benefit from this concentration of insects; however, whether this applies to pallid bats, which tend to concentrate their foraging near or on the ground, is unknown.

Colony sizes are variable, but maximum densities appear to be related to mid-summer densities of insect prey (Hermanson and O'Shea 1983). As discussed previously in Foraging Requirements, pallid bats often feed on ground insects, which may make them more vulnerable to injury and predation (Hermanson and O'Shea 1983).

Population Status and Trends

Global: Secure (NatureServe 2011)

State: Vulnerable (CDFW 2013)

Within Plan Area: Same as state

Pallid bat is a California Species of Special Concern, but little data is available to assess population status and trends. Ellison et al. (2003) compiled 292 observations for 133 colonies in 11 western states, including 35 (12%) from California. About 35% of the observations were from Arizona, 18% from Oregon, and 10% from New Mexico. However, most (78%) of the observations were collected before 1990. Information from only two sites was adequate to assess population trends: a bridge roost in Arizona that declined from 80 individuals to zero and a decline in a colony using crevices in cliffs in the Verde Valley of Arizona concurrent with increases in human activity in the area (Ellison et al. 2003). In California, Miner and Stokes (2005) noted a serious decline of pallid bats in the South Coast Ecoregion, especially in low-lying areas. They report that even as late as 1948 the species was considered to be abundant in buildings, but that by the 1970s only 1 of 12 known roost sites was still extant. Recent survey information for San Diego County indicates that few roosts that support bat species typically found in association with the pallid bat also include the species (Miner and Stokes 2005). Based on this

apparent population decline, Miner and Stokes (2005) concluded that pallid bats are highly intolerant of urban development.

Threats and Environmental Stressors

As a colonial roosting species, pallid bats are particularly vulnerable to disturbances of roost sites through vandalism, extermination, and destruction of buildings used as roost sites (Hermanson and O'Shea 1983), as well as to recreational activities such as rock climbing. As noted previously, a decline in an Arizona colony occurred concurrent with an increase in human activity (Ellison et al. 2003). Miner and Stokes (2005) found that pallid bats have abandoned almost all previously occupied sites in the urbanized areas of the South Coast Region since the late 1940s. Beck and Rudd (1960) observed that female pallid bats are particularly sensitive to disturbance during the period prior to giving birth through weaning. A single disturbance may cause them to abandon the maternity roost prior to giving birth or to move to a more secluded part of the roost after giving birth (Beck and Rudd 1960).

Food availability may be reduced by pesticides or habitat modification or degradation such as conversion to agriculture, prescribed fires, and wildfires. Pesticides and heavy metals also may contaminate prey, causing secondary poisoning. Because this species often forages on the ground, it is susceptible to predation by urban-related predators (e.g., cats and possibly dogs) and potentially collection or harassment by humans.

Several recent studies have documented substantial mortality of bats at wind energy facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). While, as of 2010, there have been no reported fatalities of pallid bats at wind energy facilities (e.g., Tetra Tech EC Inc. 2010), Solick and Erickson (2009) indicate that there have been relatively few systematic, post-project, bat-fatality monitoring data collected for large, wind-energy projects in the arid southwestern United States. Although fatalities of this species at wind energy facilities have not been documented, it is expected that the species could be at risk from turbine strikes, or other factors associated with turbine operation, such as barotrauma, hypothesized to cause bat fatalities at wind facilities (Cryan and Barclay 2009). Pallid bats would be at greatest risk of turbine strikes or from other associated causes if a facility was located within a few miles of a day

roost site (where most foraging activity occurs), and strikes would most likely occur during emergence and return to the day roost. Risk of strikes may also be higher during dispersal when young are leaving the natal roost site and fly in straight lines from the roost at altitudes of 80 feet or more (O'Shea and Vaughan 1977). Risk of strikes may be relatively low during foraging activities because pallid bats tend to forage on or close to the ground.

Conservation and Management Activities

Pallid bat is addressed in the West Mojave Plan (BLM 2005). Under Alternative A (the Proposed Action – Habitat Conservation Plan), BLM would implement several conservation measures for pallid bat, including:

- Protection of all significant roosts (defined as maternity and hibernation roosts supporting 10 or more individuals) by installing gates over mine entrances and restricting human access (The West Mojave Plan identified two significant maternity roosts and one significant maternity/hibernation roost for pallid bat on BLM-managed lands);
- Protection of bat roosts in the Pinto Mountains by gating known and new significant roosts and notifying claim holders on BLM lands containing significant roosts;
- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public;
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, abandoned buildings, or bridges to determine whether significant roost sites are present; and
- Safe eviction of bats at a non-significant roost (i.e., fewer than 10 individuals) prior to disturbance or removal.

In addition, as a BLM sensitive species, pallid bat is addressed under land use actions undertaken by BLM. In accordance with BLM's "6840 – Special Status Species Management" manual, the objectives for sensitive species policy are:

To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the

likelihood of and need for listing of these species under the ESA” (BLM 2008).

Under this policy, BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

Pallid bat is also addressed in the Military Integrated Resource Management Plans (INRMP) for the China Lake Naval Air Weapons Station (NAWS and BLM 2004) and the Marine Air Ground Task Force Training Command, Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in these INRMPs, pallid bat is provided protection and management considerations during the land use planning process defined in the China Lake Comprehensive Land Use Management Plan and military training operations at Twentynine Palms. If it is determined to be at risk from a proposed project or training activities, efforts are made to avoid and minimize impacts. For example, at Twentynine Palms, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition;
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant; and
- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

There are relatively few data for pallid bat in the Plan Area. As noted in Distribution and Occurrences with the Plan Area, there are only 59 data records for the Plan Area, of which 39 are recent. Although this species is considered common in the Great Basin, Mojave, and Sonoran deserts, there is little information about roost sites, particularly winter roosting sites and hibernacula. There is also little information on seasonal movements.

Management and Monitoring Considerations

The primary management and monitoring consideration for the pallid bat is protection of day and night roosts from disturbance that may cause abandonment. This species requires very specific thermal conditions in day roosts (e.g., deep crevices that provide an optimum thermal environment), plus the additional factor that day roosts tend to be near water resources. These habitat requirements likely result in relatively few highly suitable day roosting sites in the Plan Area. Any occupied day roosts, therefore, should be considered a highly valuable resource, and impacts should be avoided. Maintaining these sites will require protecting them from human disturbances and adjacent land uses that could cause direct mortality or injury of pallid bats or abandonment of the roost site.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for pallid bat, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 19,196,457 acres of modeled suitable habitat for pallid bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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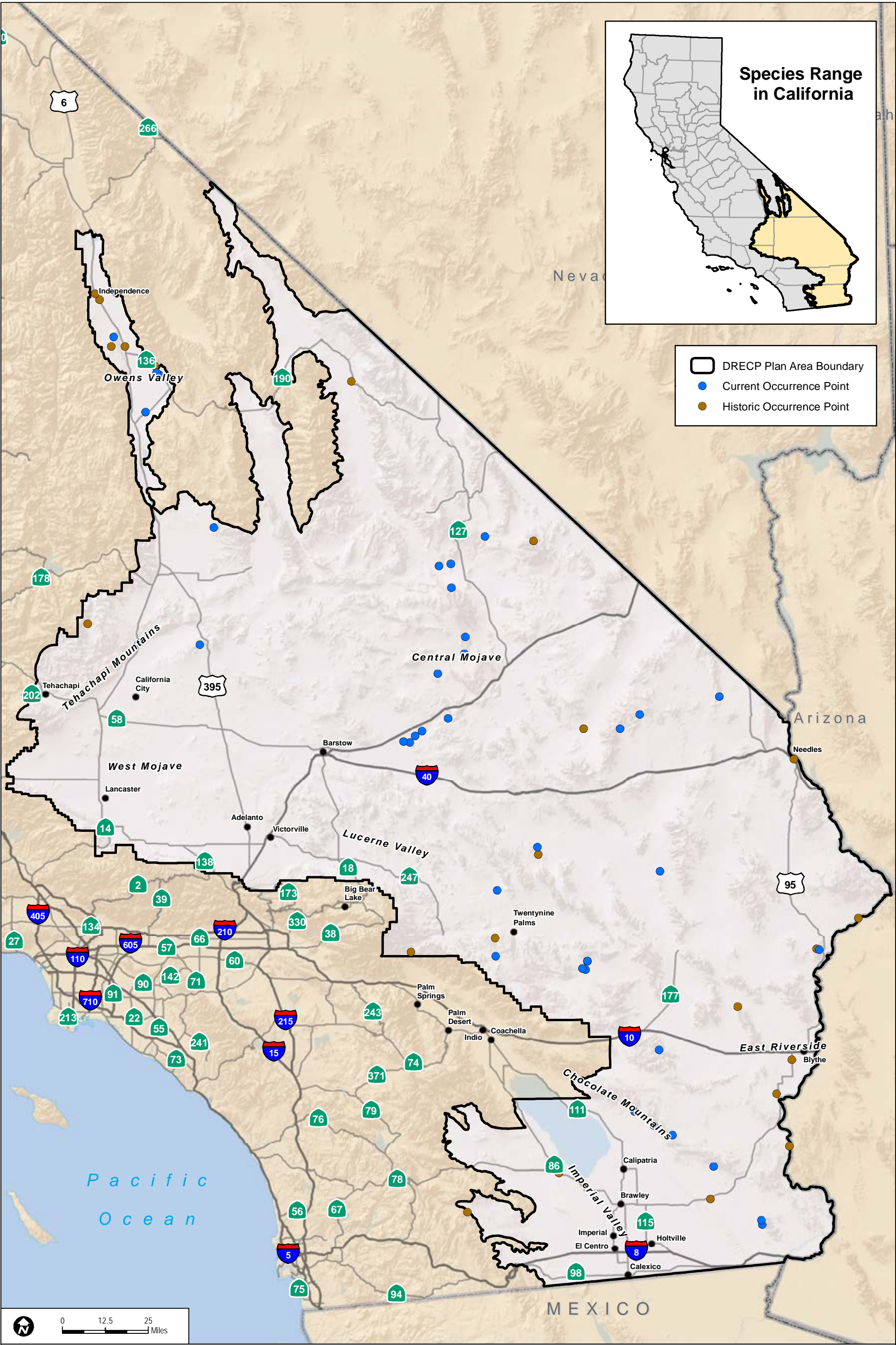
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Townsend's Big-Eared Bat (*Corynorhinus townsendii*)

Legal Status

State: Species of
Special Concern

Federal: Bureau of Land
Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Rob Schell Photography.

Taxonomy

The taxonomy of Townsend's big-eared bat (*Corynorhinus townsendii*) has undergone some recent revisions. Although the species was originally assigned to the genus *Corynorhinus* (Hall 1981), Handley (1959) reassigned it to the genus *Plecotus*, based on physical measurements, with *Corynorhinus* placed in a subgenus. More recent phylogenetic work using physical characters (Frost and Timm 1992; Tumlison and Douglas 1992) and mitochondrial DNA analysis (Hoofer and Van Den Bussche 2001) have resulted in *Corynorhinus* being restored to a separate genus within the plecotine bats.

There has also been past uncertainty in California about the distinction and distributions of two subspecies: *C. t. townsendii* and *C. t. pallescens* (see discussion in CDFG 1998). While the two subspecies occur in geographically discrete locations, their distributions have been recently revised based on mitochondrial DNA, with *C. t. townsendii* occurring throughout western and southwestern Canada and *C. t. pallescens* generally limited to New Mexico and Colorado (Piaggio et al. 2009). There are areas of sympatry in Colorado where the two subspecies are not genetically different (Piaggio et al. 2009), but based on genetic information, the subspecies in California and the Desert Renewable Energy Conservation Plan (DRECP) Area is *C. t. townsendii*. Nonetheless, in California the full species *Corynorhinus townsendii* is designated a Species of Special Concern, so the subspecific distinction in the distribution of *C. t. townsendii* and *C. t. pallescens* is not critically important for planning purposes. The species' physical characteristics are described in detail in Kunz and Martin (1982).

Distribution

General

The Townsend's big-eared bat ranges throughout the western United States; British Columbia, Canada; and Mexico (Kunz and Martin 1982). In the United States, it occurs in a continuous distribution in all of the western states and east into western South Dakota, northwestern Nebraska, southwestern Kansas, western Oklahoma, and western Texas (Piaggio et al. 2009). This continuous distribution comprises three subspecies: *C. t. townsendii*, which based on the recent genetic data (Piaggio et al. 2009) has the largest distribution range from Canada south into Mexico; *C. t. pallescens*, which is primarily limited to Colorado and New Mexico; and *C. t. australis*, which occurs in southwestern Kansas, western Oklahoma, western Texas, and north-central Mexico (Piaggio et al. 2009). The other two subspecies occur in disjunct distributions: *C. t. ingens* in southeastern Kansas, northeastern Oklahoma, southwestern Missouri, and northwestern Arkansas; and *C. t. virginianus* in eastern Kentucky, West Virginia, and Virginia (Piaggio et al. 2009).

Within California, Townsend's big-eared bat occurs throughout the state, with the exception of alpine and subalpine areas of the Sierra Nevada (Figure SP-M07), although they have been found in the subalpine zone in the White Mountains to the east of the Sierra (Szewczak et al. 1998).

Distribution and Occurrences within the Plan Area

Historical

Townsend's big-eared bat may occur throughout the Plan Area, but there are relatively few documented large maternity and/or hibernation roosts. A comprehensive review of the species' distribution was conducted by Pierson and Rainey (CDFG 1998) based on a review of historical records and field surveys conducted from June 1987 to January 1991. Their review included portions of the Plan Area known to support substantial populations, including the Owens Valley and areas east of the Sierra Nevada Range in Inyo County, the Providence Mountains in San Bernardino County, and the lower Colorado River area in San Bernardino, Riverside, and Imperial counties (see Figure 1 in CDFG 1998). They surveyed all known

maternity colonies with at least 30 individuals. Most of the active large maternity roosts within or near the Plan Area were in abandoned mines east of the Sierra Nevada range and the western slopes of the White Mountains bordering the Owens Valley. Active maternity roosts were also found in the Kingston Range area of eastern Inyo County, the Providence Mountains in northeastern San Bernardino County, and along the lower Colorado River in eastern Riverside County. An active maternity roost and a hibernation roost were also found in east San Diego County. No longer active roosts (i.e., previously known roost sites) or roosts made unavailable by human activities (e.g., inappropriate gating) were found in the Coso Range area of southern Inyo County, a site in the Providence Mountains, and two sites along the Lower Colorado River in Riverside and Imperial counties, respectively (see Figure 1 of CDFG 1998). As of 1991, Pierson and Rainey (CDFG 1998) estimated 11 active sites east of the Sierra Nevada (including several sites north of the Plan Area and the site in the Kingston Range) totaling about 1,300 adult females, 1 site in the high desert totaling about 75 adult females, 1 site in the lower desert totaling about 50 adult females, and the 2 east San Diego County sites with an unknown number of adult females. Pierson and Rainey (CDFG 1998) indicate that no large hibernation sites have been found in the desert regions of California and that smaller hibernation sites (5 to 20 individuals) are more typical of the desert; these sites are not included in the data reported by Pierson and Rainey. The lack of documented large hibernation sites in the Plan Area may reflect a lack of extensive exploration of mines and caves at higher elevations where they would more likely hibernate (CDFG 1998). However, because it is unlikely that mines and caves in the Plan Area, which are at lower elevations, have subsurface temperatures low enough for hibernation (i.e., less than 10 degrees Celsius [50.0 degrees Fahrenheit]) (see discussion in Habitat Requirement), additional exploration for hibernation sites may be irrelevant (Szewczak, pers. comm. 2012).

The DRECP database for Townsend's big-eared bat, comprising Bureau of Land Management (BLM) and California Natural Diversity Database (CNDDDB) (CDFW 2013; Dudek 2013) records, includes 13 historical records (pre-1990) for the Plan Area, dating from 1914 to 1983, as well as one record with an unknown observation date. An additional 8 records are from areas within 5 miles of the Plan Area boundary. These data generally accord with the information provided

in Pierson and Rainey (CDFG 1998), with clusters of occurrences in the southern Owens Valley–eastern Sierra Nevada area, especially the mountain ranges north of Ridgecrest. Historical records are also known from the Providence Mountains, the Kingston Range, the lower Colorado River, and Hesperia north of the San Bernardino Mountains.

See Figure SP-M07 for current and historical occurrences of Townsend's big-eared bat in the Plan Area.

Recent

There are 39 recent (i.e., since 1990) records in the Plan Area and 42 additional records within the 5-mile buffer area around the Plan Area (CDFW 2013; Dudek 2013). The geographic areas of the recent occurrences are similar to the historical occurrences, with clusters of observations in the Owens Valley–eastern Sierra Nevada area, Providence Mountains, and the Kingston Range. There is also a cluster of recent occurrences north of Barstow and along the northern slopes of the San Bernardino Mountains. There are relatively few recent occurrences from the lower Colorado River, consistent with the information reported by Pierson and Rainey (CDFG 1998).

As with the historical data, the specificity of these recent occurrence data is variable, with some records identifying roosts and others only including general location information for observations. This dataset, therefore, should be viewed as reflecting the recent documented distribution of the species in the Plan Area and should not be used as detailed data for specific roosts sites.

Natural History

Habitat Requirements

Townsend's big-eared bat is primarily associated with mesic habitats characterized by coniferous and deciduous forests and riparian habitat, although it also occurs in xeric areas (Kunz and Martin 1982). In California, this species was historically associated with limestone caves and lava tubes located in coastal lowlands, agricultural valleys, and hillsides with mixed vegetation. The species also occurs in man-made structures and tunnels (Kunz and Martin 1982), mines (López-González and Torres-Morales 2004), and the basal hollows of old-growth redwood trees (*Sequoia sempervirens*) on the north coast of California

(Gellman and Zielinski 1996; Zielinski and Gellman 1999). Within the Plan Area, Townsend's big-eared bat is primarily associated with mines in the California desert and also largely associated with man-made structures, tunnels, caves, and the basal hollows of old-growth redwood trees. In a study in northern Utah, caves and mines were the most frequently used type of roosts. More than 84% of roosts were in caves, and more than 21% of abandoned mines were used as day roosts; notably, no bridges were used (Sherwin et al. 2000). Occupied day roosts typically were subject to little disturbance by humans. Maternity colonies tended to be located in large complex sites with multiple openings (Sherwin et al. 2000). It has been suggested that the Townsend's big-eared bat has become more common in the western United States due to the availability of man-made structures (Kunz and Martin 1982); however, see discussion under Population Status and Trends. Many roosting sites in the California coastal area are in buildings, but in the Plan Area most roosting sites appear to be in abandoned mines (CDFG 1998).

Unlike many cave-roosting bat species, Townsend's big-eared bat only roosts in the open, often hanging from walls and ceilings (CDFG 1998). In the summer maternity roosts, females roost in the warm parts of caves and buildings in clusters (Kunz and Martin 1982). The census of maternity roosts in California found an overall mean colony size of about 112 individuals (CDFG 1998), which is larger than generally reported in the literature (e.g., Kunz and Martin 1982). Males appear to roost solitarily near the maternity roosts. In winter, roosting occurs solitarily or in small clusters, and Townsend's big-eared bat may share hibernacula with other bat species (Kunz and Martin 1982) (see Ecological Relationships). This species may require relatively cold temperatures to hibernate (Humphrey and Kunz 1976). Townsend's big-eared bats roost in relatively cold parts of caves in well-ventilated areas near entrances, but may move to more temperate parts of the cave if temperatures become too cold (e.g., subfreezing) (Clark et al. 2002; Humphrey and Kunz 1976; Kunz and Martin 1982) (also see discussion under Spatial Activity).

Pierson and Rainey (CDFG 1998) provide detailed information for the physical features of roosting sites in California, which is summarized below. The reader is directed to the Pierson and Rainey report for more detailed information.

Pierson and Rainey (CDFG 1998) examined potentially suitable and accessible caves, tunnels (e.g., old mine workings, water diversion tunnels, and abandoned railroad tunnels), abandoned and little-used buildings, and older (pre-1960) bridges throughout California. Censuses of bats at occupied roosts were based on direct counts or estimates for an area covered by a cluster of bats. The physical characteristics of roosts described as follows are summarized from Pierson and Rainey (CDFG 1998).

As of 1998, maternity roosts were distributed among the different structures as follows: 23 (43%) in caves; 21 (39%) in mines; 8 (15%) in buildings; and 2 (4%) in other structures (an abandoned bridge and a diversion tunnel). All roosts could be classified structurally as "cave analogues" that contained a relatively large, but enclosed space with a substantial opening. All but one of the roost entrances ranged from at least 15 centimeters (5.9 inches) in height and 31 centimeters (12.2 inches) in width, with the smallest being 15 centimeters (5.9 inches) high and 46 centimeters (18.1 inches) wide. The one exception was a mine roost in which the opening was about 10 centimeters (3.9 inches) high and 60 centimeters (23.6 inches) wide. All roosting sites were at least 1 meter (3.3 feet), and usually 2.5 to 5.0 meters (8.2 to 16.4 feet) off the ground. All roost sites were classified as semi-dark to dark settings. Mean temperatures of maternity roosts and roosts occupied by single individuals and small clusters were not significantly different. The mean temperature of maternity sites was 24.1 degrees Celsius (75.4 degrees Fahrenheit), and the mean temperature of sites with individuals and small clusters was 22.2 degrees Celsius (72.0 degrees Fahrenheit). The temperature range for maternity sites was typically 18 to 30 degrees Celsius (64.4 to 86.0 degrees Fahrenheit), but was measured as low as 14 degrees Celsius (52.2 degrees Fahrenheit). Roost relative humidity was not a factor, but tended to be relatively dry on average at about 33% (range 19 to 93%).

Assessing and characterizing hibernacula was more difficult than maternity sites because individuals tend to move among different sites during a hibernation season (CDFG 1998; Sherwin et al. 2003). Similar to maternity roosts, hibernacula are typically caves, or cave analogues, but differ in often being L-shaped, with vertical and horizontal entrances that generate a "cold sink" with significant air flow. Consistent with the literature for the species, hibernacula used in California often represent the coldest non-freezing temperature

available. In the northern counties of Shasta, Siskiyou, and Lassen, where individuals probably hibernate longer periods of time, mean hibernating roost temperature was 4.3 degrees Celsius (39.7 degrees Fahrenheit). In warmer regions of coastal and Southern California, individuals arouse periodically during the winter and occur in warmer hibernacula. The mean hibernaculum temperature for known sites throughout California is 7.1 degrees Celsius (44.8 degrees Fahrenheit)), and preferred hibernating temperatures are always below 10 degrees Celsius (50.0 degrees Fahrenheit) (CDFG 1998).

Townsend's big-eared bats forage for insects in a variety of habitats, primarily between the canopy and mid-canopy of forests, woodlands, and riparian zones, but also in sagebrush shrubsteppe (Fellers and Pierson 2002). Fellers and Pierson (2002) noted that Townsend's big-eared bats avoided foraging in grasslands. As discussed below in Spatial Activity, most foraging occurs in relatively close proximity to the day roost.

Potential roosting and foraging habitat associations for Townsend's big-eared bat in the Plan Area are provided on Table 1.

Table 1. Habitat Associations for Townsend's Big-Eared Bat

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Abandoned mines	Day roosts	TBA	CDFG 1998
Woodland, forest, riparian, desert wash	Foraging	Woodland, forest, riparian, desert wash within 6.2 miles of day roosting habitat	Fellers and Pierson 2002

Foraging Requirements

Several studies in various parts of the Townsend's big-eared bat's range found that Lepidoptera (moths) are its primary prey, including in the southwest (Ross 1967), eastern and western Oregon (Whitaker et al., 1977, 1981), and Virginia (Sample and Whitmore 1993). In Oregon, big-eared bats feed almost exclusively on moths (Whitaker et al. 1977, 1981). In Virginia, moths comprised about 90% of the species' diet by volume and percentage, followed by Coleoptera (beetles), Diptera (flies), and Hymenoptera (bees and wasps), and

reflected the abundance of these orders in interior forests (Sample and Whitmore 1993).

Reproduction

Reproduction by Townsend's big-eared bats in California is fairly well known, based on a study by Pearson et al. (1952), described herein (Table 2). Breeding begins in autumn, with peak breeding in November through February. Females store the sperm until ovulation in the spring, which may occur during and after females leave hibernation. Upon leaving hibernation, females form maternity colonies in the late spring and early summer; males during this period appear to roost singly (CDFG 1998). Gestation varies from 8 to 14 weeks, depending on degree of torpor and spring temperatures. Females have one pup. In California, birth occurs in the late spring to early summer over a 3- to 5-week period beginning in late May. Although young are born fairly undeveloped, they grow rapidly and reach adult body proportions (i.e., forearm length) in 1 month. They are capable of flying in 2.5 to 3 weeks and are weaned by 6 weeks. Both males and females are reproductive in their first autumn. Immediate postnatal mortality is about 4% to 5%, and 3-year survival is 70% to 80% for adults and 38% to 40% for yearlings (i.e., survival increases with age) (Kunz and Martin 1982).

Female maternity groups are stable and faithful to roost sites that may be used by several generations (CDFG 1998). Females remain in the natal group while males disperse after their first summer (CDFG 1998). Maternity roosts begin to break up in August.

Table 2. Key Seasonal Periods for Townsend's Big-Eared Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	x	x								x	x	x
Birth/ Development				x	x	x	x	x				
Male Dispersal								x	x			
Hibernacula	x	x	x	x	x					x	x	x

Source: Pearson et al. 1952.

Spatial Activity

Pierson and Rainey (CDFG 1998) characterize Townsend's big-eared bat as "quite sedentary" because marked animals (all females) moved no more than a few kilometers from their natal roost. Also, most activity outside of day roosts (e.g., foraging, night roosting) occurring relatively close to the roost (CDFG 1998). Recorded maximum distance from the day roost in California is 32.2 kilometers (20.0 miles) and 64.4 kilometers (39.9 miles) in Kentucky (Kunz and Martin 1982). Average distance from maternity roosts to winter hibernacula is 11.6 kilometers (7.2 miles) (range: 3.1 to 39.7 kilometers [1.9 to 24.6 miles]) (Kunz and Martin 1982). Based on a personal communication from Pearson, Pierson and Rainey (CDFG 1998) noted that when maternity colonies disband in the fall, a banded individual had never been recorded at hibernacula more than 43 kilometers (27 miles) from the banding site. However, there is also indirect evidence that Townsend's big-eared bats can travel much longer distances than indicated by direct observations of foraging activity and movement between maternity roosts and hibernacula, based on telemetry and banding studies. The genetic work by Piaggio et al. (2009) indicated gene flow by dispersing males in Colorado has occurred between roost sites 310 kilometers (192 miles) apart.

Nightly movements for bats in Marin County, California, were monitored using radiotelemetry by Fellers and Pierson (2002). Bats typically traveled less than 10.5 kilometers (6.5 miles) from the day

roost, and most flight was in the immediate vicinity of native vegetation where foraging was assumed to occur, and particularly along the edges of riparian vegetation. Similarly, on Santa Cruz Island off the coast of California, foraging activity occurred in native forest habitat within 5 kilometers (3.1 miles) of the day roost (Brown et al. 1994). Nightly foraging tended to occur in the same areas at the Marin County site (Fellers and Pierson 2002), but a study in Oregon shows shifts in foraging areas over time related to changes in prey availability (Dobkin et al. 1995). Clark et al. (1993) found that Ozark big-eared bats (*C. t. ingens*) selected foraging habitats non-randomly in relation to their availability, with edge habitats along streams and on mountain slopes used more frequently. In the Marin County study, females generally traveled greater distances than males for foraging, with their centers of activity 3.2 ± 0.5 kilometers (2.0 ± 0.3 miles) from the roost, compared to 1.3 ± 0.2 kilometers (1.1 ± 0.1 miles) for males (Fellers and Pierson 2002). Fellers and Pierson (2002) note, however, that commuting distances and patterns of nighttime activity are likely to be quite variable in relation to factors such as individual differences, sex, season, reproductive condition, and available suitable foraging habitat. For example, females may travel farther from the maternity roost or be more active foraging away from the roost later in the reproductive season when young are more independent and resources are needed to support lactation. Clark et al. (1993, 2002) found that Ozark big-eared bat nightly activity changed relative to birth and maturation of young, with nighttime returns to the maternity roost more frequent when young were totally dependent on the mother, and farther foraging distances by adult females as young matured.

Although fidelity to maternity roosts is high, there may be little fidelity to roost sites at other times of the year, possibly in relation to availability. In Oregon, there was little fidelity to night roosts in the period between emergence from hibernacula and use of maternity sites, possibly because in this study area the lava flow topography provided numerous roost sites (Dobkin et al. 1995). It is expected that use of different roost sites is locally variable in relation to roost availability.

Townsend's big-eared bats are considered to be a hover-gleaner forager based on wing morphology (Norberg and Payner 1987, as cited in Fellers and Pierson 2002), and they are agile and maneuverable fliers. They have low wing loading and high lift capacity (Kunz and Martin 1982). Fellers and Pierson (2002) found that most flight was at

10 to 30 meters (33 to 98 feet) above ground between the mid-canopy and canopy of trees. Flight through grassland was fast and low to the ground, indicating that bats were not foraging in grasslands.

Spatial activity within roosts sites likely reflects behavioral thermoregulatory adjustments. During hibernation, individuals arouse frequently and change position or move to more temperate areas of the hibernaculum (Kunz and Martin 1982). Disturbances may also cause movements within roosts sites.

Ecological Relationships

Townsend's big-eared bats may share hibernacula with other bat species; in the eastern United States, it has been found in association with Rafinesque's big-eared bat (*C. rafinesquii*) and in the western United States with big brown bat (*Eptesicus fuscus*), cave myotis (*Myotis velifer*), western small-footed myotis (*M. ciliolabrum*), dark nosed small-footed myotis (*M. melanorhinus*),¹ and California myotis (*M. californicus*) (Kunz and Martin 1982), but there is no evidence in the literature of direct competitive or symbiotic relationships with other bats. Congregations with other bat species at both day and night roosts may simply reflect use of limited resources.

With regard to potential resource partitioning, Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce inter-specific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-, within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering.

Although Townsend's big-eared bat has been characterized as a "relatively late flyer" by Kunz and Martin (1982), there are numerous observations that individuals leave roosts promptly at dusk like other species (Szewczak, pers. comm. 2012). Further, there is no information to suggest resource partitioning or direct competition for prey with other species. Although, artificial lighting may affect competitive predator-prey relationships among some bats (e.g., Frank 1988;

¹ Both *M. ciliolabrum* and *M. melanorhinus* were once considered subspecies of *M. leibii*, which is the species listed in Kunz and Martin (1982), but Wilson and Reeder (2005) list both as distinct species.

Longcore and Rich 2004), the potential for this occurring in Townsend's big-eared bats is low because this species roosts and forages away from human-developed areas (Szewczak, pers. comm. 2012).

Population Status and Trends

Global: Apparently secure (NatureServe 2011)

State: Vulnerable to imperiled (CDFG 2011)

Within Plan Area: Same as state

Townsend's big-eared bat is a California Species of Special Concern, but there are little systematic data to quantitatively assess population status and trends (e.g., numbers of individuals). However, past studies have shown a broad-ranging decline in the species through large parts of its range in the western United States (i.e., mainly the *C. t. townsendii* and *C. t. pallescens* subspecies). Human disturbance has eliminated most historical roosting sites in California and all known previously occupied limestone caves in the state have been abandoned (see discussion in Threats and Stressors). The census by Pierson and Rainey (CDFG 1998) in California, conducted from 1987 to 1991, found substantial population declines over the previous 40 years, with a 52% loss in the number of maternity colonies, a 44% decline in the number of available roosts, a 55% decline in the total number of animals (primarily adult females), and a 32% decrease in the average size of remaining colonies. Fate of roosts sites was related to the type of roost, with 88% of roosts in buildings no longer available, and 50% of roosts in caves and 57% in mines no longer used. Pierson and Rainey (CDFG 1998) also reviewed population information for other western states as of 1998, summarized below.

- Arizona – 13 verified maternity roosts, representing 10 separate colonies, with a total population of about 1,000 adult females. Two cave populations extirpated and another declined by 50% in 2 years after its cave roost was commercialized. Another population historically supporting several hundred adult females numbered fewer than 100 individuals.
- Colorado – hibernaculum with more than 500 individuals in December 1968 apparently reduced to only a few animals. Only four maternity sites had been documented in Colorado since 1970, and the largest had only approximately 80 adult females.

- New Mexico – >10,000 individuals hibernating in a timber-lined 100-meter-deep mine shaft in 1992. The shaft was burned by vandals, and several hundred dead animals were seen still hanging from the walls, and thousands more were presumed dead.
- Idaho – surveys of known hibernating sites indicate a 60% population decline since 1987.
- Nevada – surveys conducted in the late 1980s to late 1990s in 96,000 km² of northeastern Nevada revealed only two small maternity sites.
- Oregon/Washington – severe population declines for both summer and winter populations in Oregon and Washington have been well documented. Known sites in Oregon and Washington contained approximately 2,700 and 800 adult females, respectively.

The isolated populations of *C. t. ingens* and *C. t. virginianus* are considered to be in danger of extinction because of their susceptibility to human disturbance (Kunz and Martin 1982), and both subspecies were federally listed as endangered in 1979 (44 FR 69206–69208).

Threats and Environmental Stressors

Townsend's big-eared bats are very sensitive to human disturbances, and a single disturbance of a maternity roost or hibernation site may cause abandonment (Zeiner et al. 1990; Kunz and Martin 1982). All known limestone cave sites in California, for example, have been abandoned (Zeiner et al. 1990). Sherwin et al. (2000) found that occupied day roosts were typically subject to little human disturbance. As discussed in Population Trends and Status, there has been a significant decline in occupied Townsend big-eared bat roosts in California. The primary cause for the observed declines was determined to be human disturbance of roosting sites (CDFG 1998). As of 1998, 37 known maternity colonies had a total population of approximately 4,250 adult females, but only three of these colonies were considered adequately protected. Declines were also indicated at four important hibernacula for which past population data were available (CDFG 1998). The selection of relatively cold parts of caves near entrances and where there is good ventilation during hibernation makes Townsend's big-eared bats sensitive to human disturbance (including deliberate vandalism and extermination) during a period when they would be least likely to respond quickly.

Also, they tend to hang from ceilings and walls in exposed parts of roosts, making them more susceptible to disturbance (CDFG 1998). It is important that hibernacula be protected from human disturbance because animals can be aroused from hibernation and forced to use fat stores necessary for hibernation.

Pierson and Rainey (CDFG 1998) provided specific information for threats to roosts in the Plan Area. The active roosts in mines on public lands in the eastern Sierra area were considered to be at risk from recreation, mine closure for hazards, and reactivation of old mining claims. An occupied mine at the China Lake Naval Air Weapons Station was vandalized in 1988 and has not been since reoccupied. Other mines have shown evidence of extensive recreational use. Even the colony at Death Valley National Monument was vandalized in 1993, greatly reducing the number of individuals using the site. In the Providence Mountains, the Mitchell Caverns colony located in the State Park was excluded from using the site in 1970 when a bat-proof gate was installed, but replacement of the gate in 1993 resulted in rapid reoccupation. Reactivation of mining in Macedonia Canyon has excluded the species, but individuals appeared to relocate to another mine. In the Colorado River Basin and eastern Mojave Desert, Townsend's big-eared bat was once common at many mine sites, and three maternity sites were known, including the Alice Mine with the largest known colony (>1,000 individuals) in California. Surveys in 1990 and 1992 found only one small maternity site in 1990 but none in 1992. Abandoned mines in this region are subject to intensive recreation, but other apparently undisturbed mines also were unoccupied. Pierson and Rainey (CDFG 1998) suggest the agricultural conversion has reduced foraging habitat and that pesticides may be affecting this species in the region.

Several recent studies have documented substantial mortality of bats at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). Despite fairly extensive monitoring, with many documented fatalities of other bat species (primarily migrant species), as of 2004, no Ozark or Virginia big-eared bats had been known to be killed at wind facilities (or at communications towers) (Johnson and Strickland 2004). In 2010, TetraTech also reported no documented fatalities of Townsend's big-eared bats at wind facilities (TetraTech EC Inc. 2010). A general review of the wind facility-related literature also failed to reveal evidence for, or discussions of, Townsend's big-eared

bat fatalities or assessed risks at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009; Cryan and Brown 2007; Johnson and Strickland 2004; Johnson and Erickson 2008; Kuvlesky et al. 2007; Piorkowski and O'Connell 2010). Nonetheless, the U.S. Fish and Wildlife Service (USFWS) has expressed concern about the potential for fatalities of the endangered Virginia big-eared bats from wind facilities in the eastern United States as they move between caves (e.g., see Johnson and Strickland 2004). Big-eared bats in the Plan Area similarly could be at elevated risk of turbine strikes or other associated causes (e.g., barotrauma) if a wind facility were located within a few miles of a day roost site (where most foraging activity occurs), and strikes would most likely occur during emergence, return to the day roost, or when seeking a night roost between bouts of foraging. Risk of strikes may also be higher when bats are moving between maternity roosts and hibernacula in the fall and spring and when young are dispersing from the maternity roost in late summer.

Conservation and Management Activities

Townsend's big-eared bat is addressed in the West Mojave Plan (BLM 2005). Under Alternative A (the Proposed Action – Habitat Conservation Plan), BLM would implement several conservation measures for Townsend's big-eared bat and other bat species, including:

- Protection of all significant roosts (defined as maternity and hibernation roosts supporting 10 or more individuals) by installing gates over mine entrances and restricting human access. The West Mojave Plan identified two significant maternity roosts and two significant hibernation roosts for Townsend's big-eared bat on BLM-managed lands.
- Protection of bat roosts in the Pinto Mountains by gating known and new significant roosts and notifying claim holders on BLM lands containing significant roosts.
- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public.
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, and abandoned buildings or bridges to determine whether significant roost sites are present.

- Safe eviction of bats at a non-significant roost (i.e., less than 10 individuals) prior to disturbance or removal.

BLM would also conduct monitoring and adaptive management for Townsend's big-eared bat. Monitoring actions include:

- Determining bat numbers in all significant roosts
- Conducting periodic surveys in the northern part of the planning area with high potential for containing significant roosts
- Determining and reporting the effectiveness of mitigation measures providing for safe exit of bats
- Reporting take from approved projects that impact bats under to California Department of Fish and Game (CDFG) and USFWS
- Monitoring population numbers using bat houses if installed.²

Adaptive management measures include:

- Gating mines where new significant roosts are found
- Installing bat houses in locations, where appropriate, if populations decline or are threatened³
- Case-by-case review of newly detected significant roosts near open routes within riparian and desert wash habitat. Corrective actions would be taken within the foraging habitat if the new roosts are impacted by open routes or new routes would be established to avoid the habitat.

In addition, as a BLM sensitive species, Townsend's big-eared bat is addressed under other land use actions undertaken by BLM. In accordance with the BLM's "6840 – Special Status Species Management" manual, the objectives for sensitive species policy are:

To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the likelihood of and need for listing of these species under the ESA (BLM 2008).

^{2,3} The independent scientific reviewer for this profile (J. Szewczak, pers. comm. 2012) indicates that bat houses would not typically provide suitable habitat for Townsend's big-eared bat because this species requires space, not cervices. An artificial roost would have to be a cave-like structure or a building-size roost.

Under this policy BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

Townsend's big-eared bat is also addressed in the Military Integrated Natural Resources Management Plans (INRMP) for the China Lake Naval Air Weapons Station (NAWS and BLM 2004) and the Marine Air Ground Task Force Training Command Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in these INRMPs, Townsend's big-eared bat is provided protection and management considerations during the land use planning process defined in the China Lake Comprehensive Land Use Management Plan and military training operations at Twentynine Palms. If it is determined to be at risk from a proposed project or training activities, efforts are made to avoid and minimize impacts. For example, at Twentynine Palms, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant
- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

Although Pierson and Rainey (CDFG 1998) conducted a thorough review of roosting sites for Townsend's big-eared bat, this information is dated. Also, in the Plan Area the current distribution and status of roosts is not well understood. For example, Townsend's big-eared bats may be using deep mine shafts that have not been accessed by qualified biologists (CDFG 1998) or monitored for bats entering or leaving (Szewczak, pers. comm. 2012).

Management and Monitoring Considerations

The primary management and monitoring consideration for Townsend's big-eared bat is protection of day and night roosts from disturbance that may cause abandonment. This species is very

sensitive to human disturbance because it tends to roost at the entrances of caves and may be found hanging from ceilings and walls where it is susceptible to disturbance. Occupied maternity and winter roosts should be considered a highly valuable resource, and impacts should be avoided. Maintaining these sites requires protecting them from human disturbances and adjacent land uses that could cause direct mortality or injury of big-eared bats or abandonment of the roost site. Protection of riparian habitats and desert wash near roost sites (e.g., within 5 miles) is also important because these areas are important prey resource areas.

Another consideration for Townsend's big-eared bat for monitoring and management is that their echolocation signals are relatively weak. (Their large pinnae amplify weak echoes from their low amplitude calls, which enable them to more closely approach their primary prey of moths, many of which can hear, and defensively react, to bat echolocation calls [Szewczak, pers. comm. 2012]). O'Farrell and Gannon (1999) found that the big-eared bat was more effectively sampled using capture methods because their calls could only be detected at less than about 5 meters (16 feet) from the bat with the existing bat detectors. New generation acoustic detectors are more sensitive and can be deployed for long time periods, and therefore are better able to detect the species (Szewczak, pers. comm. 2012). Nonetheless, monitoring for this species may remain a challenge because the probability of detection could still be limited without broad spatial coverage of monitoring stations due to its restricted area around the primary roost used for foraging (Szewczak, pers. comm. 2012). Further, this species is difficult to physically capture due to its slow flight and high maneuverability (Szewczak, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Townsend's big-eared bat, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 16,824,190 acres of modeled suitable habitat for Townsend's big-eared bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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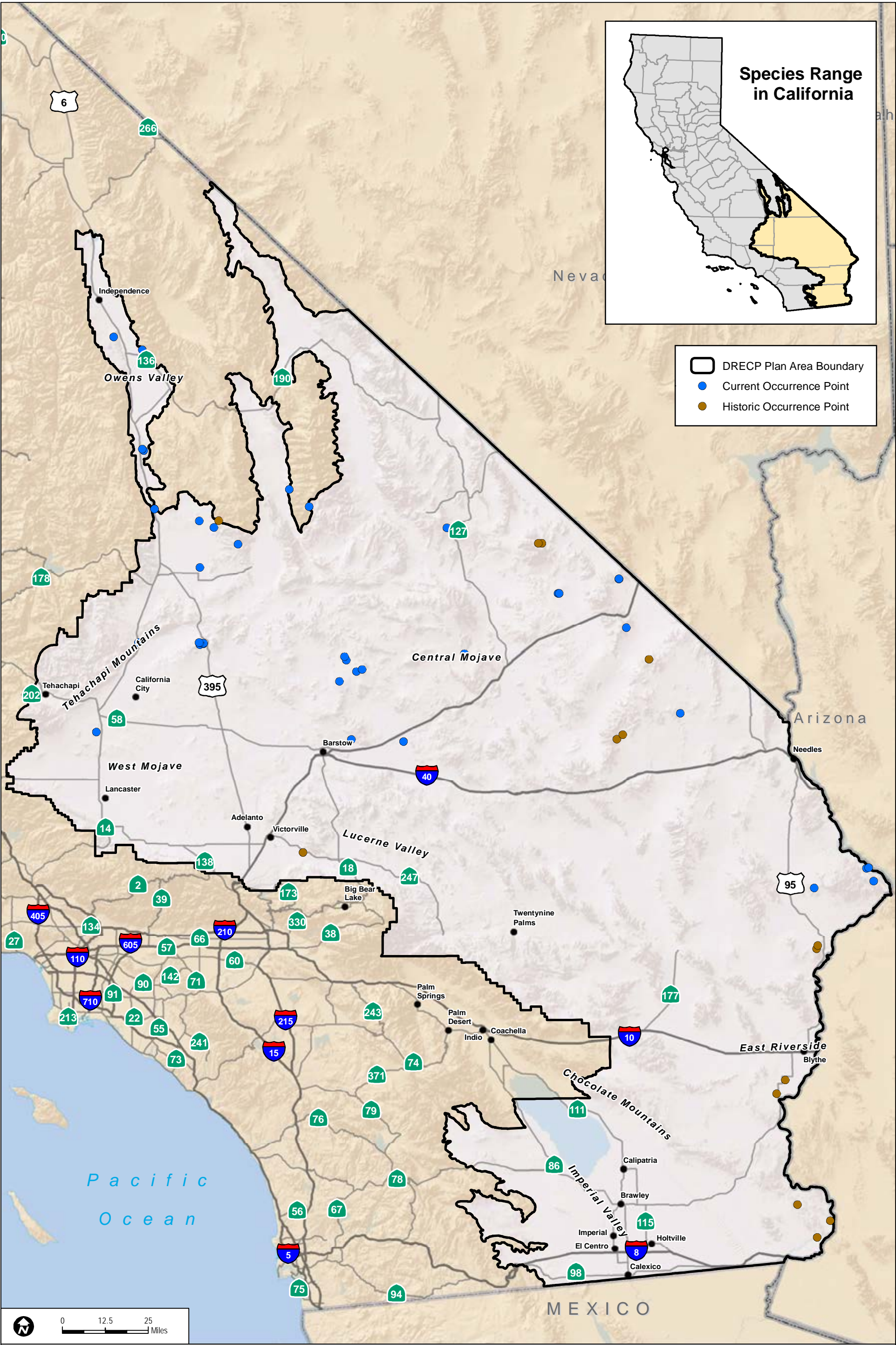
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M07
Townsend's Big-eared Bat Occurrences in the Plan Area

Alkali Mariposa-Lily (*Calochortus striatus*)

Legal Status

State: S2¹

California Rare Plant

Rank: 1B.2²

Federal: Bureau of Land Management Sensitive; U.S. Forest Service Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Dr. Heath McAllister.

Taxonomy

Alkali mariposa-lily (*Calochortus striatus*) is a perennial bulbiferous herb in the lily family (Liliaceae) (Jepson Flora Project 2011). Alkali mariposa-lily was described by S.B. Parish in 1902 (IPNI 2011). Although it appears that alkali mariposa-lily has been uniformly accepted as distinct since 1940, it was once considered by some to be synonymous with *C. palmeri* based partly on confusion of type specimens (Greene and Sanders 2006).

Alkali mariposa-lily stands approximately 1 to 4.5 decimeters (3.9 to 17.7 inches) in height (Munz and Keck 1968). A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011) and Greene and Sanders (2006).

Distribution

General

Alkali mariposa-lily occurs in Southern California and western Nevada (Jepson Flora Project 2011). Within Southern California, alkali mariposa-lily occurs in Tulare, Kern, Los Angeles, and San Bernardino counties (CNPS 2011). More specifically, this species occurs in southern Sierra Nevada; in the Mojave Desert; at the north

¹ **S2:** Imperiled.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly threatened in California.

base of the San Bernardino and San Gabriel Mountains; and in the southern San Joaquin Valley (Figure SP-P01; CDFW 2013a; Jepson Flora Project 2011; Munz and Keck 1968). Of the 102 total occurrences recorded in the California Natural Diversity Database (CNDDB), 87 are in the Plan Area (CDFW 2013a). It is rare in Nevada, with only three occurrences recorded (NNHP 2001).

Distribution and Occurrences within the Plan Area

Historical

Of the 294 localities documented in the Plan Area, 18 are considered historical. Localities considered historical have not been observed since 1989, or were recorded in 2005, but have been extirpated or possibly extirpated. They range from Kelso Valley southeast to Twentynine Palms with most localities at or near Edwards Air Force Base (AFB) (Figure SP-P01) (CDFW 2013a). The 276 remaining localities recorded since 1990 and presumed extant are discussed below.

Recent

The recent localities (i.e., since 1990) of alkali mariposa-lily reported in the Plan Area by the CNDDB range from Red Rock Canyon State Park southeast to Joshua Tree National Park. The majority of localities are located on or in the vicinity of Edwards AFB (CDFW 2013a). Alkali mariposa-lily populations are most concentrated in the metapopulation that ranges from Lancaster to Edwards AFB (CDFW 2013a). A total of 126 localities are located on Edwards AFB, and 120 of these are managed by the Department of Defense (DOD), while 6 are privately owned. Other public localities include two on lands managed by the Department of Parks and Recreation (DPR) at Red Rock Canyon State Park, four on lands managed by Los Angeles County, one on lands managed by the National Park Service (NPS) at Joshua Tree National Park, one on lands managed by the BLM, and 15 on lands managed by Rosamond Community Services. About 108 localities are on privately owned land and ownership is unknown for 19 localities (CDFW 2013a).

Natural History

Habitat Requirements

Alkali mariposa-lily grows in seasonally moist alkaline habitats such as alkaline meadows and seeps, and ephemeral washes, within chaparral, chenopod scrub, and Mojavean desert scrub (CNPS 2011; CDFW 2013a; Jepson Flora Project 2011). Alkali mariposa-lily grows in calcareous sandy soil (Fiedler 1985, cited in Greene and Sanders 2006). It prefers claypans and sand dunes, especially along drainages, in halophytic (associated with saline soils) saltbush scrub (Edwards AFB 2002). Periodic natural inundation is important to alkali mariposa-lily (Edwards AFB 2002), however, alkali mariposa-lily has been reported as absent from areas with surface salts or areas with permanent standing surface water (Mitchell 1988, cited in Greene and Sanders 2006). This species ranges in elevation from 224 to 5,240 feet (BLM 2010; CDFW 2013a).

Some associated species include saltgrass (*Distichlis spicata*), rushes (*Juncus* spp.), sedges (*Carex* spp.), beardgrass (*Polypogon* sp.), dock (*Rumex* sp.), alkali sacaton (*Sporobolus airoides*), beardless wildrye (*Elymus triticoides*), dwarf checkerbloom (*Sidalcea malviflora*), rabbitbrush (*Chrysothamnus* sp.), Baltic rush (*Juncus balticus*), and yellow sweetclover (*Melilotus indicus*) (CDFW 2013a). Table 1 lists primary habitat associations and parameters for the alkali mariposa-lily.

Table 1. Habitat Associations for Alkali Mariposa-Lily

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Chaparral, chenopod scrub, Mojavean desert scrub, meadows, and seeps	Primary	Calcareous sandy soils, alkaline, seasonally moist, 224 to 5,240 feet elevation	CNPS 2011; Greene and Sanders 2006; BLM 2010; CDFW 2013a

Reproduction

Alkali mariposa-lily blooms from April to June (CNPS 2011). Alkali mariposa-lilies have perfect flowers (i.e., which contain both the male and female reproductive parts) (Tollefson 1992, cited in Greene and

Sanders 2006). The plants arise from small membranous-coated bulbs. It is unknown whether reproduction is most commonly from seedling establishment or bulb division (Greene and Sanders 2006). Alkali mariposa-lily is pollinated by bees and flies (Tollefson 1992, cited in Greene and Sanders 2006). Although seed dispersal mechanisms for this species are unknown, seeds of some other species of *Calochortus* are gravity-dispersed (Miller et al. 2004).

Ecological Relationships

Other than the habitat associations and pollination by bees and flies described above, little is known of the life history and ecological relationships of alkali mariposa-lily.

Abundances of alkali mariposa-lily fluctuate substantially from year to year (NatureServe 2011). The bulb remains dormant and may not sprout in dry years, and the bulb may not compete well since the species is not found in stands of tall grasses (Greene and Sanders 2006).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2009)

State: S2, Imperiled (CDFW 2013b)

Abundance figures are complicated by large fluctuations from year to year, making population trends difficult to assess (NatureServe 2011). Despite its relatively wide distribution, the majority of the populations are small with the exception of the metapopulation that ranges from Lancaster to Edwards AFB (CDFW 2013a). A majority of the species' known occurrences are within California, with the exception of several occurrences in western Nevada.

At Red Rock Canyon in the Plan Area there were 44 plants reported in 1988, 13 in 1989, 133 in 1990, and 1,200 in 2003 (CDFW 2013a).

There are as many as 165,000 plants in 67 areas documented on Edwards AFB (Greene and Sanders 2006). Approximately 3,641 plants were observed in the center colony in 1995. Outside of Edwards AFB, approximately 400 plants were reported at three sites around Lancaster in Los Angeles County in 1988, but this likely

represents an underestimate of the population of alkali mariposa-lily in this area (Greene and Sanders 2006). In San Bernardino County, 50 to 100 plants were reported in 1982 at Box “S” Springs; fewer than 50 were reported at the edge of Cushenbury Springs in 1981; 30 to 40 plants were seen at Rabbit Springs in 1980; approximately 1,500 plants were reported in 1989 at Paradise Springs; and 2 plants were observed north of Paradise Springs in 1989 (CDFW 2013a). Also in San Bernardino County, fewer than 1,000 individuals were seen at Joshua Tree National Park in 2004 (CDFW 2013a).

Threats and Environmental Stressors

Alkali mariposa-lily is threatened by urbanization, grazing, trampling, road construction, hydrological alternations, and water diversions that lower the water table (CNPS 2011). It is also threatened by military operations, dumping, and grading (NatureServe 2011).

The greatest threat to alkali mariposa-lily is the lowering of water tables, which alters the seasonally moist alkaline habitat that this species requires. Urbanization in the Lancaster area is likely the second most severe threat to this species since the largest populations are concentrated near Lancaster (CDFW 2013a; Greene and Sanders 2006). Large populations along Sierra Highway that are primarily on private land and receive minimal protection are in danger of extirpation from expanding urbanization from Lancaster (CDFW 2013a; Greene and Sanders 2006).

Road construction also threatens this species. Historically, extirpations or population declines occurred with construction of Highway 18 at Whiskey Springs in the 1920s; with the expansion of Kaiser Cement, now Mitsubishi Cement Corp., in 1988 that included diking the flow of the spring and adding a parking lot at Cushenbury Springs; and with the development of a site with 300 plants near Radio Tower Meadow in 1989 (Greene and Sanders 2006; Deacon 2007).

Trampling and grazing may also severely reduce alkali mariposa-lily's reproductive capacity. A survey around Lake Isabella found that plants in ungrazed areas were taller, more robust, and more numerous than those in cattle grazed areas. From 1984 to 1991 low-intensity horse grazing was tested at The Nature Conservancy's Kern River Preserve to determine the effect that soil disturbance and

reduction of competing grasses and weeds would have on alkali mariposa-lily productivity. The grazed alkali mariposa-lily population did not experience a substantial increase or decrease compared to non-grazed control populations under low-intensity grazing (Tollefson 1992, cited in Greene and Sanders 2006). Pavlik et al. (2011) also documented strong impacts by mammalian herbivores on alkali mariposa-lily growth and reproduction in two consecutive years at Ash Meadows National Wildlife Refuge.

Although it may not be a more widespread problem, ongoing monitoring at The Nature Conservancy's Kern River Preserve suggests that competition from taller grasses, such as beardless wildrye (*Elymus triticoides*) and non-native barley (*Hordeum* spp.), may contribute to population declines (Tollefson 1992, cited in Greene and Sanders 2006).

Conservation and Management Activities

Thirty-nine alkali mariposa-lily occurrences are recorded on the Edwards AFB (CDFW 2013a). The Edwards Air Force Base Integrated Natural Resources Management Plan offers general conservation measures based on an ecosystem approach with a general goal of conserving and improving the habitat that would benefit all native species (Edwards AFB 2002). One of the goals included in the Plan is to review project plans to ensure drainage patterns are not changed in areas where listed or sensitive species, such as alkali mariposa-lily, occur (Edwards AFB 2002). Populations at the Nature Conservancy's Kern River Preserve populations are currently protected from development (Greene and Sanders 2006). Additional populations are on public and private lands with unknown conservation and management activities.

Data Characterization

Population trends are difficult to assess due to the large year-to-year fluctuations (NatureServe 2011). Some key components of the life history of the species have not been characterized. The most common mode of reproduction is not known. In addition, seed dispersal mechanisms are not known. However, because there is information available for other similar species of *Calochortus*, and because there is recent occurrence information available for this species, there is sufficient information available to characterize this species.

Management and Monitoring Considerations

Because population numbers fluctuate widely year to year, alkali mariposa-lily requires long-term monitoring to detect population trends. Possible measures to maintain or restore the water table at its historic level and to remove or modify existing obstructions to natural spring or seep flows would benefit the species and should be discussed with land managers. Trampling and grazing by cows should be prevented by fencing known population sites. Although it has yet to be tested for this species, control of introduced weeds could reduce competition for resources, and thus improve reproductive capability (Greene and Sanders 2006). Protection from herbivores is essential for achieving stable or increasing population trends (Moore, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for alkali mariposa-lily, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 188,549 acres of modeled suitable habitat for alkali mariposa-lily in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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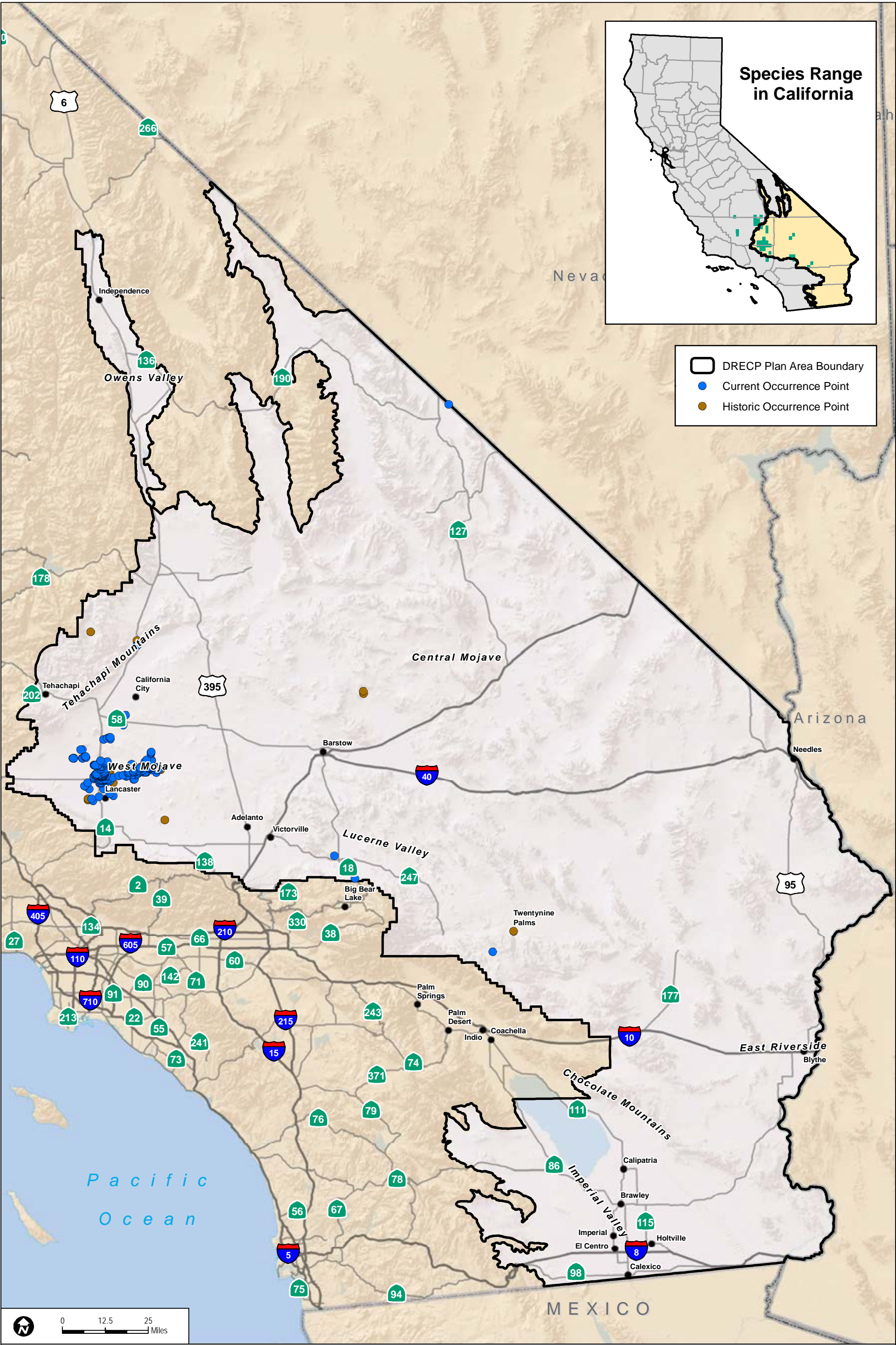
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PLANTS

Alkali Mariposa-Lily (*Calochortus striatus*)

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P01
Alkali Mariposa Lily Occurrences in the Plan Area

Bakersfield Cactus (*Opuntia basilaris* var. *treleasei*)

Legal Status

State: Endangered, S2.1¹

California Rare Plant

Rank: 1B.1²

Federal: Endangered, U.S. Forest Service Sensitive

Critical Habitat: N/A

Recovery Planning: Recovery Plan for Upland Species of the San Joaquin Valley, California (USFWS 1998)

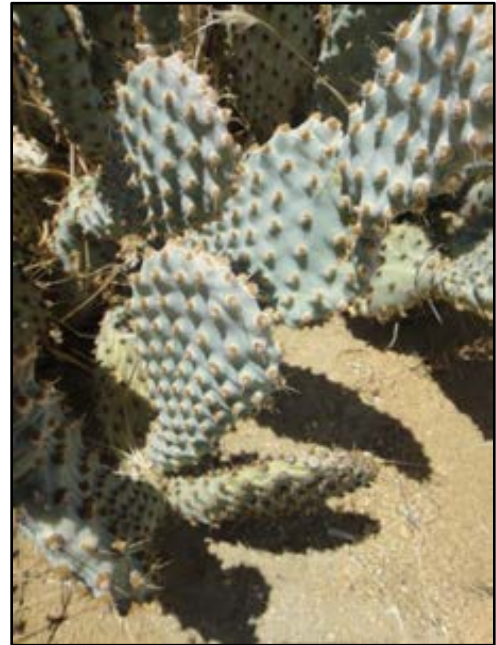


Photo courtesy of Neal Kramer.

Taxonomy

Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is a perennial stem succulent in the cactus family (Cactaceae) (Jepson Flora Project 2011; CNPS 2011). Bakersfield cactus was originally published as *Opuntia treleasei* by J.M. Coulter in 1896 (IPNI 2011). Bakersfield cactus was listed as *Opuntia treleasei* in the Federal Register notice announcing the endangered status of the species (55 FR 29361–29370). Bakersfield cactus has been consistently treated as a variety of *Opuntia basilaris* in every major California flora, including Munz and Keck (1959), Munz (1974), Hickman (1993), FNA(1993), and Baldwin et al. (2012), is to treat Bakersfield cactus as a variety of *O. basilaris* since the publication of Jepson's 1936 *A Flora of California*.

Bakersfield cactus is low growing with stem segments approximately 9 to 20 centimeters (3.5 to 7.9 inches) long (USFWS 2011; Jepson Flora Project 2011). A full physical description of the species can be found in the Jepson eFlora (Jepson Flora Project 2011).

¹ **S2:** Imperiled; **X.1:** Very threatened.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously threatened in California.

Distribution

General

Bakersfield cactus occurs in the Tehachapi Mountain area and the southeastern San Joaquin Valley in Kern County, California (Figure SP-P02; Jepson Flora Project 2011). The historical distribution of Bakersfield cactus was likely more or less continuous east of Bakersfield, from Granite Station south to Comanche Point, east to Caliente, and west to Oildale (USFWS 1998, 2011). However, it is currently restricted to a limited area of central Kern County near Bakersfield in the southern San Joaquin Valley (USFWS 2011), and in the vicinity of Oak Creek and Mojave (Kentner, pers. comm. 2012). Approximately one-third of the historical population has been extirpated (USFWS 1998). The California Natural Diversity Database (CNDDB) includes 46 occurrences, of which 6 are in the Plan Area at 9 different localities (CDFW 2013a). However, there are a large number of records from the Plan area that were submitted to CNDDB in 2011, but have not been made publically available yet (Kentner, pers. comm. 2012; CDFW 2013a).

Following the recent discovery of the plants near Oak Creek, surveys for Bakersfield cactus were conducted on several thousand acres of proposed wind energy developments in the adjacent foothills of the eastern Tehachapi Mountains and the creosote brush and Joshua Tree woodlands of the desert areas to the east (Kentner, pers. comm. 2012).

Most of the individuals of the cactus population in this area are unambiguously identified as *Opuntia basilaris* var. *basilaris*, or beavertail cactus. However, the population is highly polymorphic and about a third of the individual plants display a varying number of morphological features that are characteristic of Bakersfield cactus (Kentner, pers. comm. 2012).

In 2010 and 2011, botanical surveys for proposed wind energy developments in the Tehachapi pass/Oak Creek area detected thousands of individual plants that were identified as Bakersfield cactus. The identification criteria were based on the recommendations of CDFG (Cypher 2011) which state that any plant with any one of several diagnostic characteristics of Bakersfield cactus should be considered to be the listed variety. Based on their identification recommendations, CDFG has been requiring Incidental

Take Permits and mitigation for the take of large numbers cactus in the vicinity of Oak Creek and Mojave within the Plan area (Kentner, pers. comm. 2012).

Point data for 1,244 individuals identified as Bakersfield cactus were submitted to CNDDDB in the summer of 2011, and surveys have been ongoing since then. However, the CDFG identification criteria are controversial, and many of the identified plants appear to be intermediate between the varieties (Kentner, pers. comm. 2012).

Distribution and Occurrences within the Plan Area

Historical

Of the nine localities documented in the CNDDDB within the Plan Area, one is considered historical with plants that have not been observed since 1934. This locality is mapped approximately 1 mile south of Fram (CDFW 2013a; Figure SP-P02). The historical locality in the Plan Area is east of the recent occurrences described below.

Recent

The eight recent localities of Bakersfield cactus reported in the Plan Area by the CNDDDB occur at Oak Creek Pass in the Tehachapi Mountains, and near West Antelope Station and east of Bean Canyon at the foothills of the Tehachapi Mountains (Figure SP-P02; CDFW 2013a). Three of these localities are located on private land; ownership of the others is unknown (CDFW 2013a). Most of these localities are all very new, found in 2009 and 2010, and extend the variety's known range southeast since they occur south of Comanche Point and east of Caliente, which were considered the range limits in 1987 according to the 5-Year Review (USFWS 2011).

Natural History

Habitat Requirements

Bakersfield cactus grows primarily in chenopod scrub, but is also found in valley and foothill grassland; and occasionally in cismontane woodland, including blue oak woodland and riparian woodland (CNPS 2011; USFWS 2011; CDFW 2013a; Jepson Flora

Project 2011). Some associated species include California filago (*Filago californica*), yellow pincushion (*Chaenactis glabriuscula*), and red brome (*Bromus madritensis* ssp. *rubens*), as well as other non-native annual grasses (USFWS 2011).

Bakersfield cactus occurs on floodplains, ridges, bluffs and low rolling hills, and flats (USFWS 2011; CDFW 2013a). Soils are sandy or gravelly with little silt and clay, are low in organic matter, and may contain cobbles or boulders (CNPS 2011; USFWS 2011); they are granitic and well-drained (CDFW 2013a). Bakersfield cactus ranges from 90 meters (295 feet) (CNPS 2011; CDFW 2013a) to 5,000 feet (Kentner, pers. comm. 2012). Table 1 lists primary habitat associations and parameters for Bakersfield cactus.

Table 1. Habitat Associations for Bakersfield Cactus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Saltbush scrub, grassland, blue oak woodland, and riparian woodland	Primary	Coarse well-drained sandy or gravelly soils, from 90 to 1,140 meters (295 to 5,000 feet) elevation	CNPS 2011; CDFW 2013a; USFWS 2011

Reproduction

Bakersfield cactus blooms from April to May (CNPS 2011).

The pollination biology of Bakersfield cactus is only relevant for the portion of the population that is genetically capable of reproduction by seed. However, that proportion remains unknown (Kentner, pers. comm. 2012). Bakersfield cactus exhibit several features that are characteristic of bee pollination: flowers are large and showy with a watermelon-like odor; it has a long flowering period; and produces large amounts of nutritious pollen from numerous stamens (Jepson Flora Project 2011; Grant and Grant 1979). Flowers of beavertail prickly-pear (*Opuntia basilaris* var. *basilaris*) are commonly visited by beetles and bees, but are pollinated mainly by bees (Grant and Grant 1979). The native solitary bee, *Diadasia australis* ssp. *california*, is a

potential pollinator of Bakersfield cactus (USFWS 2011). This bee is known to occur in Kern County and specializes in collecting pollen from prickly-pear species. *Diadasia* bees in general are oligolectic (exhibit a narrow, specialized preference for pollen sources), with some specializing on cactus species. The little cactus bee (*Diadasia rinconis*) has been recorded as a visitor to Bakersfield cactus (Grant and Grant 1979).

Chromosome counts indicate that at least some Bakersfield cactus are triploid (2 of the 3 plants that have been examined were triploid ($2n = 3X = 33$); Pinkava et al. 1977, 1992). Triploid plants are typically at least partially sterile and may have a greatly reduced capacity for sexual reproduction either via pollen or by seed. Triploid populations therefore often rely predominantly on vegetative reproduction—the production of new plants from sources other than seed. Fallen pads can take root. Cactus pads may be dispersed by flood waters. Seed dispersal agents are unknown (USFWS 2011), but the fruits and vegetative parts of *Opuntia* species in general, such as the spiny pad, are closely linked with seed dispersal and vegetative dissemination by animals (Reyes-Agüero et al. 2006). Bakersfield cactus does not survive prolonged inundation (USFWS 2011).

Morphological evidence indicates that gene flow (*i.e.* hybridization) between *O. b. basilaris* and *O. b. treleasei* may be occurring in the populations near Oak Creek. The issue of the ploidy of Bakersfield cactus is highly relevant to the question of hybridization between the varieties. Both the proportion of triploid vs. diploid individuals in Bakersfield cactus populations and the frequency with which triploid individuals produce euploid gametes that would be compatible with the gametes of diploid individuals, including *O. b. basilaris*, is currently unknown (Pinkava et al. 1977, 1992).

Ecological Relationships

Competition with non-native grasses for water is likely the cause of the decline in the number of cactus pads and low rates of reproduction observed in recent population studies at Sand Ridge Preserve (USFWS 2011).

A study conducted from 2002 to 2005 at Sand Ridge Preserve analyzed the effects of grass clipping and Fusilade II (a grass-specific

herbicide) treatments on Bakersfield cactus survival, flower production, and recruitment. Bakersfield cactus declined on the control plots, and the rates of both vegetative and sexual reproduction were low, likely due to a reduction in soil moisture storage by non-native annual grasses in years with below average precipitation. In contrast to the control plots, the number of cactus pads in the clipped plots and herbicide-treated plots increased (USFWS 2011). A decline in pollinators may be partly responsible for the low levels and infrequency of seed set observed (USFWS 2011).

Predation of Bakersfield cactus is unknown, though it is not considered to a threat to this species (USFWS 2011). In Mexico, the seed and fruits of other *Opuntia* species are consumed primarily by rodents, but also by harvester ants, birds, and other mammals (González-Espinosa and Quintana-Ascencio 1986).

Population Status and Trends

Global: G5T2, variety is Imperiled (NatureServe 2011, Conservation Status last reviewed 1990)

State: S2.1, Imperiled (CDFW 2013b)

Once likely more or less continuous east of Bakersfield, the current range of Bakersfield cactus consists of scattered fragments of these once larger populations (USFWS 2011).

Though the total population of Bakersfield cactus was not estimated historically, densely spaced clumps of cactus once covered an estimated area of 2 square miles from the Caliente Creek floodplain onto Sand Ridge (USFWS 2011). When known sites were inventoried in 1989, fewer than 20,000 clumps of Bakersfield cactus were estimated to remain. Only four areas had populations of 1,000 clumps or more: Comanche Point, Kern Bluff, Sand Ridge, and the area north of Wheeler Ridge (USFWS 2011). A status survey in 2010 and 2011 was conducted to determine the current state of the historical occurrences of Bakersfield cactus throughout its range (USFWS 2011; Cypher et al. 2011a). Based on these surveys which focused on existing CNDDDB occurrences, 25 occurrences are confirmed extant, 11 are believed to be extirpated, the status of 3 could not be determined, 2 previously unreported populations were documented, and 6

undocumented translocated populations were identified. Therefore, there is a minimum of 33 extant occurrences (Cypher et al. 2011a).

Threats and Environmental Stressors

Agricultural land conversion, oil development, sand mining, urbanization, off-road vehicle use, proposed flood control basins, telecommunication and electrical lines construction, and possibly wildfires were considered threats to Bakersfield cactus habitat at the time of its listing in 1990 (USFWS 2011). Currently, the loss and modification of habitat from agricultural conversion, wind energy development, and urban, especially residential, development remain the largest threats to Bakersfield cactus (USFWS 2011; Kentner, pers. comm. 2012). Threats today also include oil development, off-road vehicle use, sand mining, and competition from non-native grasses. In addition, climate change, air pollution (including elevated nitrogen deposition), loss of pollinators, flooding, and loss of genetic diversity have been identified as potential new threats (USFWS 2011). However, loss of genetic diversity is not relevant to the unknown proportion of the population that is triploid and undergoing clonal reproduction (Kentner, pers. comm. 2012).

Conservation and Management Activities

A recently-completed survey has provided updated information on the status of known occurrences, confirming at least 33 current occurrences (Cypher et al. 2011a).

In 1990, The Nature Conservancy doubled the size of the Sand Ridge Preserve to 270 acres by acquiring a remnant of the Caliente Creek wash at the eastern base of the ridge. In 1997, the preserve was transferred to the Center for Natural Lands Management (USFWS 2011; CNLM 2011).

Since 1993, with implementation of the Metropolitan Bakersfield Habitat Conservation Plan, several colonies of Bakersfield cactus have been acquired. The Implementation Trust for the Metropolitan Bakersfield Habitat Conservation Plan has protected parts of occurrences within the Kern Bluffs and Sand Ridge recovery sites (USFWS 2011). Negotiations over the proposed Department of Water Resources (DWR) Habitat Conservation Plan (HCP) for the California

Aqueduct right-of-way are currently stalled with no target date for HCP completion (Grunewald 2011).

The approximately 100,000-acre Wind Wolves Preserve at the very southern end of the San Joaquin Valley is owned and run by the Wildlands Conservancy. There are approximately 50 acres of presumed occupied Bakersfield cactus habitat on the Wind Wolves Preserve within the Wheeler Ridge recovery site (USFWS 2011).

Tejon Ranch Corporation negotiated with national conservation groups on a preservation agreement, executed on June 17, 2008, in which Tejon Ranch Corporation committed to placing aside 178,000 acres through a combination of dedicated and designated project open spaces and allowing the conservation organizations to purchase up to an additional 62,000 acres at State-appraised cost. The conservation easement established through the agreement would result in the permanent conservation of almost 90% of the Ranch (USFWS 2011).

The California Native Plant Society (CNPS) transplanted Bakersfield cactus clumps from sites proposed for development to Sand Ridge Preserve and the California Living Museum in Bakersfield. In addition, a few of the cactus clumps growing on the East Hills Mall site in Bakersfield were removed prior to mall construction, then replanted in a display bed after construction. No monitoring of transplanted individuals has occurred at any of the sites to determine survival rates or reproductive success (USFWS 1998). Hundreds if not thousands of Bakersfield cactus plants have been relocated during the construction of wind energy developments near Oak Creek and Mojave. Relocations there are ongoing (Kentner, pers. comm. 2012).

Data Characterization

Distribution of Bakersfield cactus is not well known. It likely occurs in additional locations that have not been documented considering there is a lot of potential habitat that has not been surveyed, primarily because this habitat occurs on private land (Cypher et al. 2011). The recent expansion on the range to include the eastern Tehachapi Mountains from recent occurrences found on wind energy development project sites in Oak Creek and Mojave has not become publically available through the CNDDDB at this time (Kentner, pers. comm. 2012).

Although inferences can be made from other *Opuntia* species, the reproductive biology of Bakersfield cactus has not been studied directly (USFWS 2011).

Management and Monitoring Considerations

The USFWS 5-year review identified the following five highest priority actions to be implemented over the next 5 years to achieve progress toward recovery (USFWS 2011):

1. Protect populations within Bakersfield City limits in the Kern Bluff area and south of Highway 178
2. Work with willing landowners to establish a conservation easement or fee title to the property at the mouth of Kern Canyon
3. Complete the draft Department of Water Resources Habitat Conservation Plan
4. Conduct census of known populations and monitor the reproductive status of known populations
5. Determine suitable management methods for reducing non-native annual grasses and increasing native perennials, including Bakersfield cactus, and communicate the benefits of such management to rangeland landowners.

Cypher et al. (2011b) translocated Bakersfield cactus pads and clumps from the Center for Natural Land Management's Sand Ridge Preserve to Kern County's Bena Landfill Conservation Area as part of a trail population establishment. Ten clumps and 25 shed pads were translocated in fall 2009. Cypher et al. (2011b) concludes that translocation may constitute an effective strategy for establishing new populations of Bakersfield cactus, but suggests continued monitoring of the success of the Bena Landfill population.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Bakersfield cactus, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements,

which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are approximately 3,421 acres of modeled suitable habitat for Bakersfield cactus in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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Barstow woolly sunflower (*Eriophyllum mohavense*)

Legal Status

State: None

California Rare Plant

Rank: 1B.2¹

Federal: Bureau of Land
Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A

Notes: In 1993, the U.S. Fish and Wildlife Service (USFWS) determined that proposing to list Barstow woolly sunflower as endangered or threatened may have been appropriate, but sufficient data on biological vulnerability and threat were not available at that time to support a proposed rule (58 FR 51144–51199).



Photo courtesy of Xeric Specialties.

Taxonomy

Barstow woolly sunflower (*Eriophyllum mohavense*) was originally described by Ivan Murray Johnston in 1923 under the synonym *Eremonanus mohavensis* (Johnston 1923; IPNI 2005), but soon included in *Eriophyllum* by Jepson (1925, p. 1117). Barstow woolly sunflower is in the sunflower family (Asteraceae) (Jepson Flora Project 2011). It is an annual herb standing approximately 1 to 2.5 centimeters (0.4 to 1 inch) in height. A full physical description of the species can be found in The Jepson Flora Project (2011) and Munz (1974).

Distribution

General

This species is endemic to California's Mojave Desert (Jepson Flora Project 2011). Barstow woolly sunflower is restricted to a range within a 30-mile radius of Kramer Junction in San Bernardino and Kern Counties. The eastern-most extant location is Barstow, while the

¹ **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly threatened in California.

westernmost is the town of Mojave, southernmost is El Mirage, and the northernmost is 25.8 mi northeast of Kramer Junction between Almond Mountain and Black Hills (CDFW 2013a). The species' elevation range extends from 2,000 to 3,600 feet (CDFW 2013a). All of the 67 total California Natural Diversity Database (CNDDDB) occurrences (at 168 localities) are in the Plan Area (Figure SP-P03).

Distribution and Occurrences within the Plan Area

Historical

There are 168 total CNDDDB localities in the Plan Area, approximately 22% (37) of which have been recorded prior to 1990 (CDFW 2013a). Additional occurrences of Barstow woolly sunflower have been extirpated without having been updated in the CNDDDB (MacKay, pers. comm. 2012). The historic occurrences extend from the area around Barstow northwest to the Almond Mountains foothills, west to the area around Kramer Junction, and south to Stoddard Mountain (CDFW 2013a).

Recent

The majority of the 134 CNDDDB localities recorded since 1990 are located in the vicinity of Kramer Junction on Edwards Air Force Base. Known extant occurrences now extend farther west, approximately 5.5 miles east of the Mojave Airport, and near Buckhorn Lake about 1 mile north of the Kern–Los Angeles County line. New records farther east are from near Opal and Lane Mountains, as well as Barstow (Figure SP-P03). The El Mirage CNDDDB occurrence, entered in November 2011, is now the known southernmost occurrence. Of the current localities, approximately 30% are on lands owned by the Department of Defense (DOD) on Edwards Air Force Base, 10% are on Bureau of Land Management (BLM) land, 6% are on lands managed by the CDFW in the West Mojave Desert, and 54% are on lands that are privately owned or are likely privately owned (CDFW 2013a).

Natural History

Habitat Requirements

Barstow woolly sunflower has been observed in openings within chenopod scrub, Mojavean desert scrub, creosote bush scrub, and also occurs on playas (CNPS 2011; Jepson Flora Project 2011). This species has been observed on bare areas with little soil that frequently contain a shallow subsurface caliche layer (BLM 2005) (Table 1). Barstow woolly sunflower often grows in the sandy margins of small “scalds”, which are slightly depressed areas (within the preferred vegetation types) with poor drainage that collect water and then evaporate. However, further away from the Kramer Junction/Edwards Air Force Base areas, it has been reported growing under different edaphic conditions. For example, the easternmost CNDDDB location is on a cobbly ridge, north-facing slope, and the occurrence at Opal Mountain is on upland gravelly soil (CDFW 2013a; MacKay, pers. comm. 2012). A 1995 study by the consulting firm, TetraTech, showed that this species tends to occupy soils with more clay in upper layers, higher alkalinity, more boron, and soil of harder consistency than adjacent unoccupied areas (cited in Andre).

Table 1. Habitat Associations for Barstow Woolly Sunflower

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Chenopod scrub, Mojavean desert scrub, Creosote bush scrub, and Playas	Primary habitat	2,000–3,600 feet	CNPS 2011; CDFW 2013a

Reproduction

Barstow woolly sunflower is a very small annual plant. Duration of flowering is from two to three weeks during the flowering period from March or April to May. Plants then generally go to fruit in May (CNPS 2011; Jepson Flora Project 2011). An 8-year study by Jim Andre in the 1990s showed that seedlings are only established in years of average or

above average precipitation. The study also showed that populations tend to occupy the same places when they do germinate, possibly indicating that there is very limited seed dispersal distance in this species (Andre and Knight 1999). Plants were successfully established off site as mitigation for the Luz solar field project (MacKay, pers. comm. 2012). There is no information available regarding pollinators.

Ecological Relationships

Very little is known about the ecological relationships of Barstow woolly sunflower. Annual species that are most frequently found with Barstow woolly sunflower in the same microhabitat include Mojave spineflower (*Chorizanthe spinosa*) and yellow pepper-grass (*Lepidium flavum*). Mojave spineflower was reported as an associated species in over half of the CNDDDB records and yellow pepper-grass was an associated species in several records as well (CDFW 2013a).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)

State: S2.2, Imperiled (CDFW 2013b)

The 2012 CNDDDB includes 63 occurrences for this species, although this estimate includes occurrences that are historic (prior to 1990) or possibly extirpated (CDFW 2013a). Population trends for this species are unknown at this time, but a multi-year, population-level study is underway by BMP Ecosciences and estimated to conclude in 2015. This is an annual plant with populations that fluctuate greatly (by orders of magnitude) from year to year depending on conditions, and also which have a soil seed bank that also likely shows a remarkable amount of fluctuation. Barstow woolly sunflower responds to water availability in terms of population dynamics (Andre and Knight 1999).

Threats and Environmental Stressors

Threats to Barstow woolly sunflower include military activities, energy and subdivision development, sheep grazing, exotic plant species, off-road vehicle use, highway and road improvements and building, mining, dumping, and pipeline construction (NatureServe 2010; CNPS 2011; MacKay, pers. comm. 2012). Of these threats, those

of primary concern include energy development, military activities, sheep grazing, off-road vehicles, and highway improvements (NatureServe 2010; MacKay, pers. comm. 2012). Energy development includes not only construction of solar and wind power production sites, but also utility corridor construction (e.g., roads, transmission lines) (MacKay, pers. comm. 2012).

Specific effects of energy development include shading from solar panels. Shading can reduce the density of Barstow woolly sunflower by suppressing emergence from the seed bank. In addition, shading from solar panels may kill plants before they flower, thus reducing seed production (Tanner et al. 2014). Shading from solar panels can also decrease species richness (i.e., the number of different species present) and community abundance (i.e., the number of individual plants present) (Tanner et al. 2014).

Several Barstow woolly sunflower sites may be extirpated, but their status has not been reported to the CNDDDB; however, it is also important to recognize that these plants may be inactive in some years but persist in the seed bank. Currently, only one CNDDDB occurrence is recorded as possibly extirpated (CDFW 2013a). However, CNDDDB Occurrences #9 and #10 occur along Highway 58 and a widening project has occurred along this highway that has likely extirpated these occurrences (CDFW 2013a; MacKay, pers. comm. 2012).

Conservation and Management Activities

The BLM has established a 314-acre botanical Area of Critical Environmental Concern (ACEC) northeast of Kramer Junction to protect the Barstow woolly sunflower in the West Mojave Plan Area. In a final West Mojave Plan EIS (BLM 2005), of which Alternative A was adopted by BLM in a March 13, 2006 Record of Decision, the protected area for Barstow Woolly Sunflower was expanded to 36,211 acres. This includes the original 314-acre fenced area (now officially called the Barstow Woolly Sunflower ACEC) plus some adjacent CDFG land (acquired by a land exchange with BLM). Along with some private inholdings, the entire 36,211 acres makes up the Barstow Woolly Sunflower Conservation Area (BLM 2005; MacKay, pers. comm. 2012). This ACEC has a perimeter fence that offers protection from human impacts. However, the BLM has little staff to police and enforce the area, so it is unclear how much protection the Barstow

Woolly Sunflower Conservation Area affords this species (MacKay, pers. comm. 2012).

Management areas at Haystack Butte and Leuhman Ridge on Edwards Air Force Base support Barstow woolly sunflower. Another management area consisting of undeveloped land north of Mercury Boulevard also supports this species (Edwards Air Force Base 2002).

Data Characterization

Little is known about the population status and ecology of Barstow woolly sunflower due to its ephemeral life history. Many of the occurrence points are relatively old and need to be updated (MacKay, pers. comm. 2012). Nearly half (29 of 63) of the CNDDB occurrences were recorded prior to 1990 or are not dated (CDFW 2013a).

Surveys seem only to be done around existing roads and trails, and especially in areas where there are proposed projects. Much more can be discovered by extensive and thorough surveys on public lands, as well as private lands (if permission granted), conducted within the flowering period and in years with average to above-average precipitation.

Management and Monitoring Considerations

Barstow woolly sunflower would likely benefit from the elimination of off-road vehicle use and sheep grazing in occupied areas. In addition, vast areas remain unsurveyed (MacKay, pers. comm. 2012). Focused surveys for this species should be conducted in suitable habitat where it is likely to occur, including investigating the status of records of the species where the status is uncertain and that may have been extirpated. Management and monitoring are complicated by the year-to-year fluctuations in population size in response to rainfall. It is very important that surveys be during the short flowering season (before fruiting) in years of average to above-average rainfall. The inadequacy of survey efforts is substantiated by the very recent 2011 discovery of Barstow woolly sunflower at El Mirage (MacKay, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Barstow woolly sunflower, using available spatial information and occurrence

information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are approximately 186,866 acres of modeled suitable habitat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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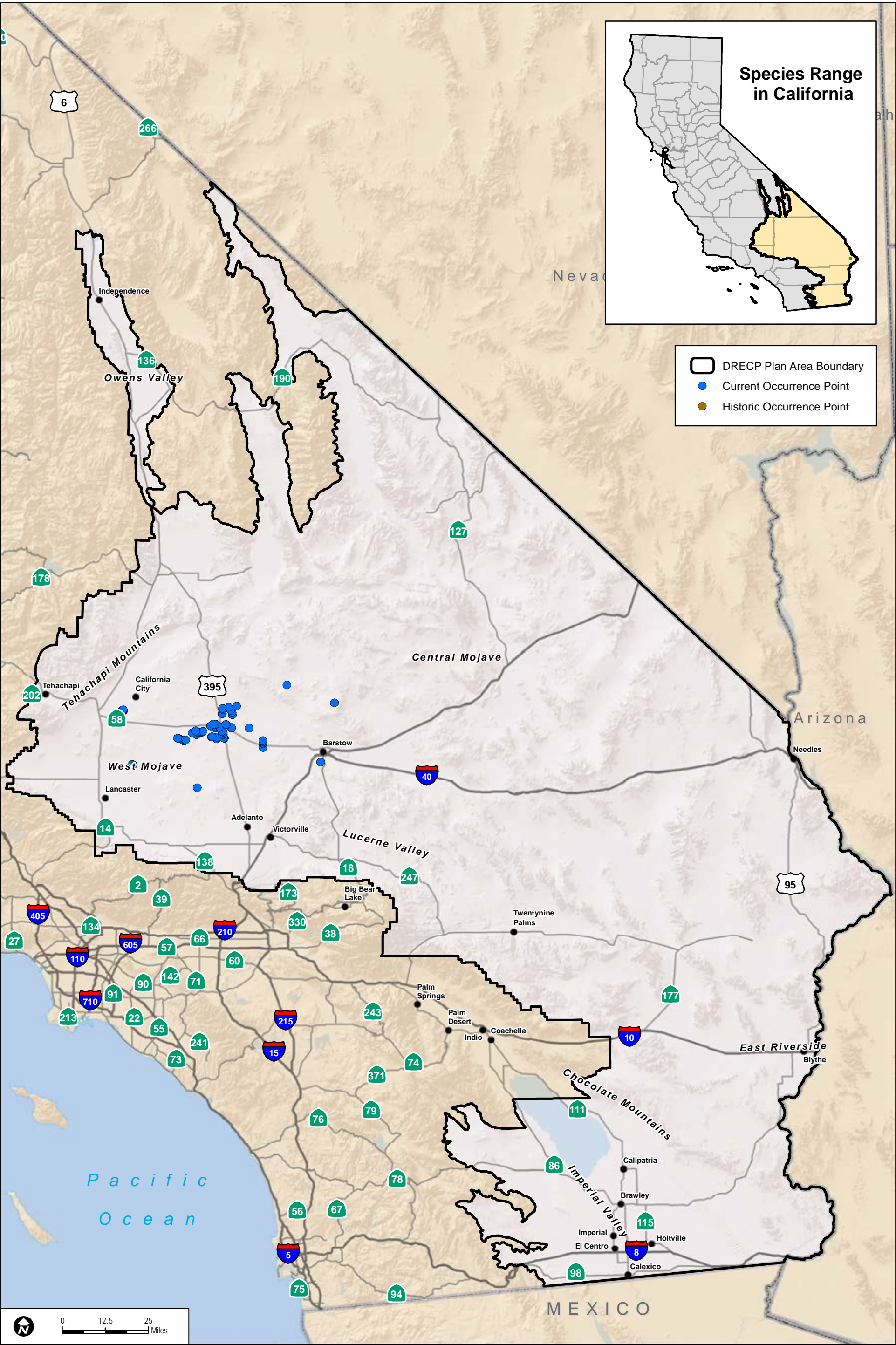
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P03
Barstow Wolly Sunflower Occurrences in the Plan Area

Desert Cymopterus (*Cymopterus deserticola*)

Legal Status

State: None

California Rare Plant

Rank: 1B.2¹

Federal: Bureau of Land
Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Jasmine J. Watts

Taxonomy

Desert cymopterus (*Cymopterus deserticola*) was originally described by Townshend Stith Brandegee in 1915 (Hall 1915, p. 168; IPNI 2005). Mathias (1930) provides a detailed description of this species, and subsequent descriptions in floras appear to be based on this work (Bagley 2006). Desert cymopterus is in the carrot family (Apiaceae) (Jepson Flora Project 2011). Desert cymopterus is a tap-rooted perennial about 15 centimeters (5.9 inches) in height. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

There are a total of 79 occurrences in the California Natural Diversity Database (CNDDB) (CDFW 2013a) all originating from 14 collections, one collection of which was a duplicate (Sanders, pers. comm. 2012). The historical distribution of desert cymopterus ranged from Apple Valley in San Bernardino County northward approximately 55 miles to the Cuddeback Lake basin in San Bernardino County, and westward approximately 45 miles to the Rogers and Buckhorn Dry Lake basins on Edwards Air Force Base in Kern and Los Angeles Counties. However, the Apple Valley locations have presumably been extirpated.

¹ **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly endangered in California.

resulting in a current distribution that includes the Rogers Dry Lake, Harper Dry Lake, Cuddeback Dry Lake, and Superior Dry Lake basins (69 FR 64884–64889; Figure SP-P04). This species occurs at elevations from 2,000 to 3,000 feet, and possibly up to 5,000 feet (69 FR 64884–64889; CNPS 2011).

Distribution and Occurrences within the Plan Area

Historical

There are three CNDDDB occurrences from before 1990. Two of these are located in the vicinity of Leuhman Ridge and Kramer Hills near other occurrences of this species. One of these is possibly extirpated and located more than 25 miles southeast of other occurrences east of Victorville (Figure SP-P04) (CDFW 2013a).

Recent

There are a total of 230 CNDDDB occurrences in the Plan Area (CDFW 2013a). Of these, there are 227 recent occurrences (status updated since 1990) that range from south of Buckhorn Lake along the Kern–Los Angeles County boundary north to the Black Hills and Fort Irwin (Figure SP-P04). However, the majority of these occurrences are located on or near Edwards Air Force Base which may be because Edwards Air Force Base is the only area in the Mojave Desert that has had extensive surveys conducted for desert cymopterus. Those on Edwards Air Force Base and the one occurrence at Fort Irwin are on lands owned by the Department of Defense (DOD). Other occurrences on public land include those managed by the Bureau of Land Management (BLM) in the general vicinity of North Edwards, Harper Lake, and Cuddeback Lake. The remaining nine recent records are either located on private land or the ownership is unknown (CDFW 2013a).

Natural History

Habitat Requirements

Desert cymopterus grows in Joshua tree woodland, saltbush scrub, and Mojavean desert scrub communities on loose, sandy soils. The sandy soils required by this species occur on alluvial fans and basins,

stabilized sand fields, and occasionally sandy slopes of desert dry lake basins (69 FR 64884–64889).

Table 1. Habitat Associations for Desert Cymopterus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Joshua tree woodland, Saltbush scrub, Mojavean desert scrub	Primary habitat	Loose, sandy soils, 2,000–5,000 feet	69 FR 64884–64889; CNPS 2011

Reproduction

As a taprooted perennial, desert cymopterus does not appear to reproduce vegetatively, but rather reproduces via seeds. Seedling establishment has not been reported for this species. Establishment of new individuals in a population may be infrequent given that many reported desert cymopterus populations are highly dispersed and low density (NatureServe 2010).

Depending on the year, desert cymopterus flowers between early March and mid-May, and may not flower at all in unfavorable years. Poor seed production or seed survival may be a factor in infrequent establishment observed in field studies. At a number of sites in several different years little or no seed production has been observed. A study conducted in 1988 at five sites found that the inflorescences dried up and aborted before setting fruit at each site (Moe 1988, cited in Bagley 2006). In a 1992 study at three sites on Edwards Air Force Base, Charlton (1993, cited in Bagley 2006) reported that only a small portion of the plants flowered and that even fewer successfully produced seed. On the other hand, in 1995, a wet El Niño year, most plants (95%) produced inflorescences at the same three sites, and 51% of the plants had set fruit near the end of the growing season (Mitchell et al. 1995, cited in NatureServe 2010). However, this still indicates a lot of inflorescences aborted before setting fruit (NatureServe 2010).

Fruits of desert cymopterus are fairly large and do not seem well adapted for dispersal over long distances. Fruits generally seem to fall relatively close to the parent plant. The fruits have a marginal wing that may facilitate dispersal by wind. However, the wings in *C. deserticola* are reduced and appear to be thickened, which suggests

that either wind dispersal is less important in this species or that the winds of the Mojave are sufficient to move seeds with poorly developed wings (Sanders, pers. comm. 2012). In addition, the fruits mature late in the season, typically after the end of the rainy season, so they remain dry and light. Therefore, given that wind is relatively common in the open sandy habitats where this species is found, it could easily push the fruits along the soil surface, although the fruits probably do not become airborne (NatureServe 2010).

Because of the annual variability in rainfall, the underground parts of herbaceous desert perennials, including desert cymopterus, must be able to maintain the populations over time with frequent years of reproductive failure; in addition, they must be able to survive prolonged periods of low soil moisture and entire years without aboveground photosynthetic activity (NatureServe 2010).

In dry years, desert cymopterus may not produce flowers or fruit and may even remain dormant underground during the usual growing season. In very wet years, however, they may produce flowers and fruits abundantly. Observations of abundant desert cymopterus in 1995 on Edwards Air Force Base demonstrated the species' ability to survive the 1988–1994 drought in large numbers and with great vigor (NatureServe 2010). Populations of desert cymopterus are probably maintained by periodic recruitment only after years of exceptionally favorable conditions for seed production (Bagley 2006; NatureServe 2010).

Ecological Relationships

Population sizes appear to vary greatly from year to year, evidently in response to the amount and timing of winter and spring rainfall, making it difficult to determine population trends (NatureServe 2010).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2005)

State: S2, Imperiled (CDFW 2013b)

Abundance estimates for each population are usually less than 1,000 plants. However, estimating population size is difficult for a number of reasons. First, occurrences and population size fluctuate widely from

year to year in response to climatic conditions, especially on the amount of rainfall. Desert cymopterus is dependent upon frequent spring rains. Furthermore, this species may remain dormant underground as a taproot and may not emerge when there is insufficient rainfall, so the number of individuals underground could be greater than the number of individuals aboveground. Also, detectability may be low in years when plants only produce leaves and no inflorescences (NatureServe 2010).

The largest and most robust populations of desert cymopterus occur on Edwards Air Force Base. Seventeen population surveys were performed during a study in 1995, a good year for the species, and population sizes at each location ranged from 1 to 1,929 individuals. In total, 14,093 individuals were counted over an area of 1,465 acres (Tetra Tech 1995, cited in NatureServe 2010).

Threats and Environmental Stressors

Desert cymopterus is potentially threatened by habitat alteration and destruction resulting from military activities on Edwards Air Force Base, the expansion of Fort Irwin, oil and gas development, utility construction, renewable energy development, off-road vehicle use, sheep grazing, Land Tenure Adjustment, and urban development (69 FR 64884–64889; CNPS 2011). However, according to the proposed rule (69 FR 64884–64889), the magnitude and relative importance of most of these potential threats were unknown. Grazing by native and non-native herbivores—presumably including mammals, insects, and desert tortoise (*Gopherus agassizii*)—is also a threat to this species. This may contribute to the low-density, dispersed nature of the majority of reported desert cymopterus populations by limiting the plants' reproductive potential and reducing their vigor (Bagley 2006).

Conservation and Management Activities

The vast majority of plants and acreage of habitat for desert cymopterus are currently thought to occur on the Edwards Air Force Base. Therefore, this species is not covered by the West Mojave Habitat Conservation Plan (Edwards Air Force Base 2002).

Management areas at Haystack Butte and Leuhman Ridge on Edwards Air Force Base support desert cymopterus. Another management area

consisting of undeveloped land north of Mercury Boulevard also supports this species (Edwards Air Force Base 2002). The Edwards Air Force Base Integrate Natural Resources Management Plan offers general conservation measures based on an ecosystem approach with a general goal of conserving and improving the habitat that would benefit all native species (Edwards Air Force Base 2002).

Data Characterization

In general, data availability for desert cymopterus is poor except for population data in some years at Edwards Air Force Base. Population trends are difficult to assess due to the fluctuations caused by variation in rainfall year to year. Furthermore, little is known regarding the species' reproduction, seed dispersal, and recruitment, and nothing is known about pollination. No studies have examined seed viability, longevity in the soil, and predation. Nothing is known of the physiology of dormancy in desert cymopterus or how long plants can survive dormancy. In addition, the requirements for seed germination and establishment of new plants in the population are unknown (NatureServe 2010).

Management and Monitoring Considerations

Protection should focus on currently known to occur on Edwards Air Force Base just south of Rogers Lake, and west and south of Leuhman Ridge. The long-term viability of populations may also rely on the protection of habitat corridors between these populations. Little is known of the distribution and abundance of desert cymopterus off Edwards Air Force Base. Focused surveys for this plant should be conducted in suitable habitat off Edwards Air Force base in favorably wet years to determine if high-density sites exist and how any such areas could be protected (Bagley 2006).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for desert cymopterus, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements,

which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 344,996 acres of modeled suitable habitat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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Little San Bernardino Mountains Linanthus (*Linanthus maculatus*)

Legal Status

State: S2¹

California Rare Plant

Rank: 1B.2²

Federal: Bureau of Land Management Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Michael Charters,
www.calflora.net.

Taxonomy

Little San Bernardino Mountains linanthus (*Linanthus maculatus*) is an annual herb in the phlox family (Polemoniaceae). The species was first described as *Gilia maculata* by S.B. Parish in 1892 from an 1889 collection at “Agua Caliente” (Palm Springs) by W.G. Wright (Jepson Flora Project 2011). During a review of the phlox family in 1904, Milliken treated this species as *Linanthus maculatus* (Milliken 1904) where it remained until the late 1980s. The species has been the subject of much controversy over the last two decades, compounded by a lack of specimens and a lack of close relatives, with Patterson (1989) concluding that the species, although unique, would best fit in the genus *Gilia*, and later Grant (1998) suggesting that the species be placed in the monotypic genus *Maculigilia*. Finally, Porter and Johnson (2000) rebutted Grant’s revision and suggested that the species should be returned to the genus *Linanthus*. The taxonomical debate over the placement of this species in *Gilia* or *Linanthus* or some other genus is unlikely to influence its current legal or conservation status.

¹ **S2:** Imperiled.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **.2:** fairly threatened in California.

Little San Bernardino Mountains linanthus is a diminutive, densely hairy, alternate-leaved annual species approximately 1 to 3 centimeters (0.4 to 1.2 inches) in height (Jepson Flora Project 2011; Patterson 1989). Descriptions of the species' physical characteristics can be found in the *Jepson eFlora* (Jepson Flora Project 2011) and in Patterson's (1989) taxonomic review of the species.

Distribution

General

Little San Bernardino Mountains linanthus is endemic to Southern California with occurrences in San Bernardino, Riverside, and Imperial counties (CNPS 2011). There are 35 collections of Little San Bernardino Mountains linanthus listed in the Consortium of California Herbaria (CCH) database (CCH 2011). The California Natural Diversity Database (CNDDB) records 39 occurrences for this species at 53 localities, but only 27 occurrences at 29 localities occur within the Plan Area (CDFW 2013a). This species' range is restricted to the mouth of Dry Morongo Canyon near the City of Desert Hot Springs and the north side of Joshua Tree National Park south of State Highway 62 in the Little San Bernardino Mountains, and from Whitewater Canyon in the eastern San Bernardino Mountains to Palm Springs. Virtually all of the Palm Springs populations are considered extirpated due to development (Sanders 2006). Additional areas where the species has been recently documented include the mouth of Rattlesnake Canyon and near the Two Hole Spring area on the northern side of the San Bernardino Mountains, and just east of the San Diego County line near Dos Cabezas Spring in Imperial County (Figure SP-P05) (CCH 2011; Sanders 2006).

Distribution and Occurrences within the Plan Area

Historical

Of the 29 localities documented in the CNDDB within the Plan Area, two localities east of Yucca Valley and west of Joshua Tree in San Bernardino County, California, is considered historical because the plants were observed once in 1937 and once in 1940, but these two localities are still presumed to be extant (Figure SP-P05) (CDFW 2013a).

Recent

The 27 recent localities of Little San Bernardino Mountains linanthus occur along the western boundary of the Plan Area in San Bernardino and Riverside counties (Figure SP-P05) (CDFW 2013a). Eight of the localities are at least partially located in Joshua Tree National Park. Seven are located on Bureau of Land Management (BLM) land in Johnson Valley, Homestead Valley, or southeastern Lucerne Valley. One is located on BLM land at the northeastern base of the San Bernardino Mountains and another is at the transition between San Bernardino and Little San Bernardino mountains (CDFW 2013a). Three are located on BLM land in Palm Canyon Wash east of San Diego County. Two localities occur on private land south of the town of Joshua Tree. The remaining five localities have unknown ownership and occur on a wash north of Joshua Tree National Park, south of State Route 62 east of Joshua Tree, at Pipes Canyon north of Yucca Valley, around Yucca Valley, and east of Yucca Valley (CDFW 2013a).

Natural History**Habitat Requirements**

Little San Bernardino Mountains linanthus grows on loose, well-aerated, open sandy benches and flats on the margins of desert washes (Sanders 2006; Jepson Flora Project 2011). It grows at 195 to 2,075 meters (640 to 6,806 feet) elevation (CDFW 2012b; CNPS 2011). A review of the elevation data from herbarium collections in the CCH (2011) indicates that the elevation range of the species is from 997 to 4,002 feet (one record indicating a collection from 20 meters elevation appears to be erroneous).

Little San Bernardino Mountains linanthus is always found in open areas that receive no shade from nearby shrubs and is associated with other small annual species, such as sigmoid threadplant (*Nemacladus sigmoideus*), blushing threadplant (*N. rubescens*), evening primrose (*Camissonia pallida*), common loeflingia (*Loeflingia squarrosa*), Arizona nest straw (*Filago arizonica*), and Wallace's woolly sunflower (*Eriophyllum wallacei*) (Sanders 2006).

Table 1. Habitat Associations for Little San Bernardino Mountains Linanthus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Desert wash ³ systems associated with desert dunes, Joshua tree woodland, and Mojavean and Sonoran desert scrub	Primary habitat	Loose sandy soils, 640 to 6,806 feet elevation	Sanders 2006; Jepson Flora Project 2011; CNPS 2011; CDFW 2013b

Reproduction

Little San Bernardino Mountains linanthus is a diminutive herbaceous annual that reproduces via seed. The ecology of Little San Bernardino Mountains linanthus is not well known because it has not been well studied, and little is known about the plant's pollinator relationships, seed viability, or seed germination (Patterson 1989; Sanders 2006; CVAG 2006). The flower is white with a vermilion spot on each spreading lobe on most individuals (Munz 1974), suggesting that the species is almost certainly insect-pollinated (Sanders 2006). The flowering time for this species is March through May (CNPS 2011). A review of the collections shows that approximately one-third of the specimens were collected in March, two-thirds in April, and only a few in February and May (CCH 2011).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed)

State: S2, Imperiled (CDFW 2013b)

There are four major populations of Little San Bernardino Mountains linanthus (Sanders 2006). All populations are extant except for the Palm Springs populations, which were located in the center of what is now Palm Springs and along Interstate 10 north of the city proper (Sanders 2006). Because of the isolated nature of desert wash systems,

³ Sanders (2006) states that populations are found only on loose sandy benches on the margins of washes... shrubs are always present in the general areas occupied, but these are not common on the sandy benches where *Gilia* actually is found.

the major populations are separated into smaller “population units” associated with individual washes (Sanders 2006). Two new populations have been discovered in the last two decades: a population in the Rattlesnake Canyon and Two Hole Spring areas on the northern side of the San Bernardino Mountains and an Imperial County population located just east of the San Diego County line near Dos Cabezas Spring (CDFW 2013a; CCH 2011).

Some estimates have been made of the number of individuals in some occurrences. About 10,000 individuals were estimated north of Indian Avenue near the mouth of Big Morongo Canyon (Riverside County) in 1996 and widespread plants observed in flat areas between Joshua Tree and Indian Cove in 1995 (G. Hemkamp, pers. comm., cited in Sanders 2006). A few hundred individuals were present in the Dry Morongo Canyon (San Bernardino County) area in 1992 and 1995 and six in 1996; and 100 plants in an area south of Joshua Tree near State Highway 62 in 1986, which were “reduced markedly” in 1987, 150–200 plants in 1988, 25–30 plants in 1990, and 1,000 plants in 1993 (Patterson 1989; CDFW 2013a).

There are several gaps in the early records for this species, including a 17-year gap from 1907 to 1924 (Sanders 2006; CDFW 2013a; CCH 2011). Only six collections were made between 1924 and 1960 and only two collections were made in the 1970s. Since the end of the 1970s, the number of collections has increased, probably because of the increase in desert botanical work and Patterson’s 1989 description of habitat for the species (Sanders 2006).

Population trends are difficult to estimate for the species because population size in a given year appears to depend on environmental conditions and fluctuates greatly from year to year.

Threats and Environmental Stressors

Little San Bernardino Mountains linanthus is potentially threatened by habitat disturbance and destruction from urban expansion, off-highway vehicle use, illegal dumping, and an increase in invasive non-native species (CNPS 2011; CDFW 2013b), and flood control activities (CVAG 2006). The largest populations are adjacent to communities, such as Yucca Valley, Joshua Tree, and Desert Hot Springs, that have grown substantially in the last two decades. Additional development

pressures associated with the expansion of these communities could impact core populations (Sanders 2006).

Flood control maintenance activities pose a specific threat to the species as these activities change the hydrological regime and sediment-carrying capacity of flows within wash systems. In particular, flood control activities pose a substantial threat to populations of Little San Bernardino Mountains linanthus in the Whitewater Canyon, Mission Creek, and Dry Morongo Canyon Wash areas (CVAG 2006).

Off-highway vehicle use is a particular threat to Little San Bernardino Mountains linanthus because the species grows only in desert washes, which are favored by off-highway vehicle users because they are so sparsely vegetated (Sanders 2006).

Conservation and Management Activities

The Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP) covers the majority of the known extant populations of Little San Bernardino Mountains linanthus. The CV MSHCP identified three “Core Habitat”⁴ areas for the species: Whitewater Canyon, Upper Mission Creek/Big Morongo Canyon, and the Morongo Wash Special Provisions Area, as well as two additional areas for conservation (CVAG 2006). Additionally, the CV MSHCP has identified approximately 3,189 acres of potential habitat for Little San Bernardino Mountains linanthus in the CV MSHCP plan area, of which approximately 2,410 acres is identified as Core Habitat. Conservation of Little San Bernardino Mountains linanthus habitat in the CV MSHCP area will amount to 2,955 acres, of which 2,235 acres, or approximately 76%, is identified as core habitat (CVAG 2006).

The CV MSHCP will result in conservation of 97% of the known occurrences of the species in the CV MSHCP plan area. Additionally, the CV MSHCP has coordinated efforts with the Coachella Valley Flood

⁴ The CV MSHCP defines Core Habitat as “The areas identified in the Plan for a given species that are composed of a habitat patch or aggregation of habitat patches that (1) are of sufficient size to support a self-sustaining population of that species, (2) are not fragmented in a way to cause separation into isolated populations, (3) have functional Essential Ecological Processes, and (4) have effective biological corridors and/or linkages to other habitats, where feasible, to allow gene flow among populations and to promote movement of large predators.”

Control District to ensure that the hydrological regime in the wash systems of conserved areas is maintained to ensure the conservation of core habitat (CVAG 2006).

The BLM West Mojave Plan (WMP) area encompasses the large population of Little San Bernardino Mountains linanthus located along the northern edge of Joshua Tree National Park in the Little San Bernardino Mountains, as well as the newly discovered populations in Rattlesnake Canyon and Two Hole Spring on the northern edge of the San Bernardino Mountains (Sanders 2006). The WMP proposes two goals and two objectives for Little San Bernardino Mountains linanthus. The goals are to: (1) protect all occurrences of the species on public lands and protect 90% of occurrences on private lands, and (2) protect drainages and the fluvial processes that define the hydrologic regimes in the wash systems. WMP objectives are to: (1) declare all occupied habitat within 100 feet of the edge of washes as Conservation Areas, and (2) limit the channelization of occupied washes (BLM 2005).

Data Characterization

Population trends for the species are difficult to determine because it appears that yearly fluctuations in population size are correlated with annual rainfall amounts.

Very little data existed for the species prior to Patterson's 1989 review of the species. Since then, much more information has been gathered and synthesized for the species, especially through the drafting of species accounts and species-specific conservation management plans under the CV MSHCP and the BLM WMP. In addition, many new populations or localities have been discovered and mapped since 1989, resulting in a greater understanding of the prime core habitat parameters for the species. Despite a general lack of knowledge on the ecology of the species (pollinator interactions, seed viability, germination requirements, etc.), it appears that enough data have been gathered to effectively draft conservation and management plans for the species.

Management and Monitoring Considerations

Future management efforts for Little San Bernardino Mountains linanthus should focus on maintaining natural unobstructed hydrological regimes in areas that support existing populations, as well as in areas with prime core habitat. This will undoubtedly entail working closely with local flood control agencies and private landowners. Additionally, increased management of off-highway vehicle use, and stricter penalties for their illegal use in areas known to support Little San Bernardino Mountains linanthus should be considered. Finally, future monitoring efforts should focus on determining population trends for known populations, as well as on identifying locations of new populations.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Little San Bernardino Mountains linanthus, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 343,289 acres of modeled suitable habitat for little San Bernardino Mountains linanthus in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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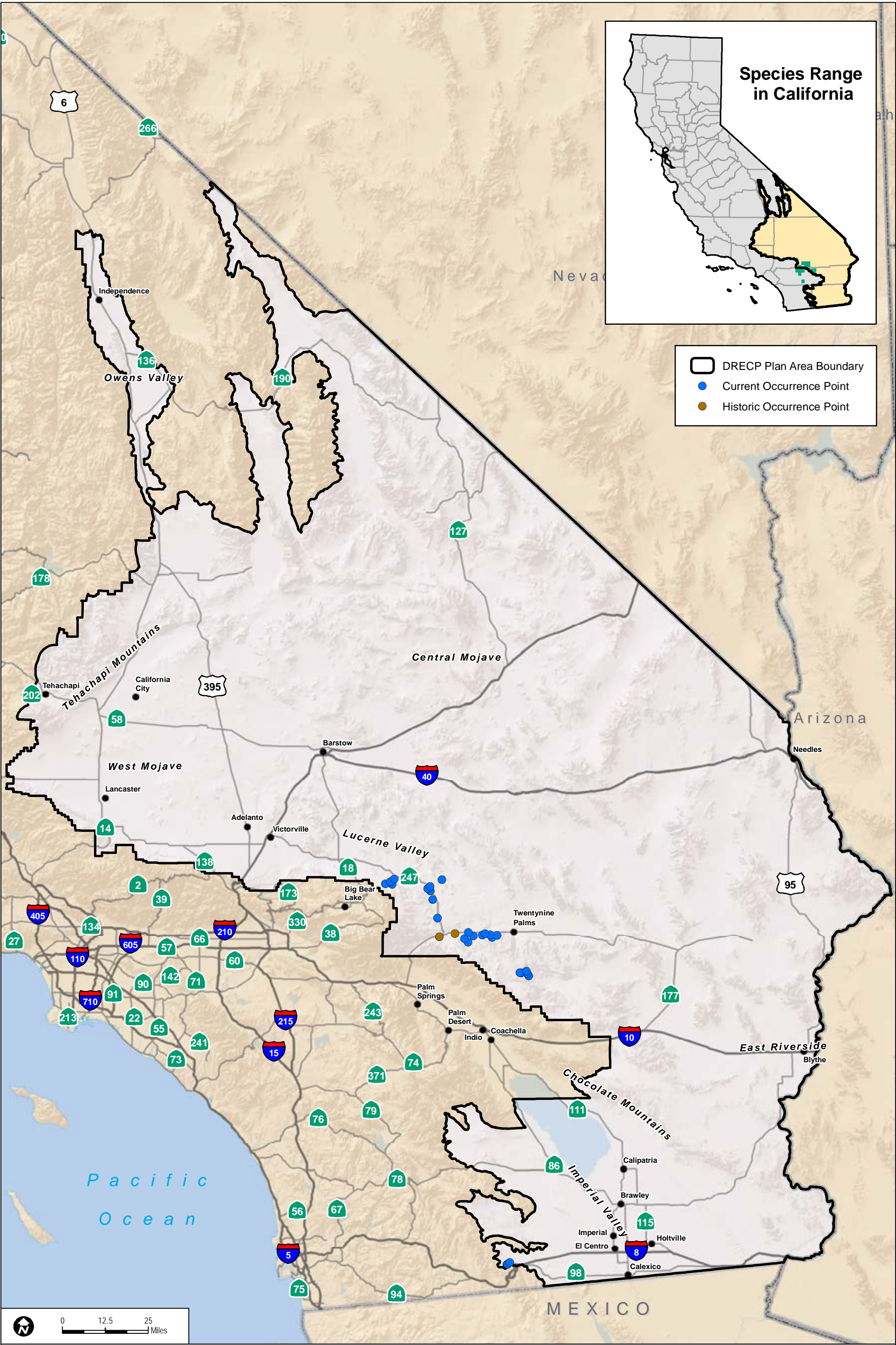
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P05
Little San Bernardino Mountains Linanthus Occurrences in the Plan Area

Mojave Monkeyflower (*Mimulus mohavensis*)

Legal Status

State: None

California Rare Plant

Rank: 1B.2¹

Federal: BLM Sensitive

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Steve Schoenig.

Taxonomy

Mojave monkeyflower (*Mimulus mohavensis*) was originally described by John Gill Lemmon in 1884 (Lemmon 1884; IPNI 2011). It is a distinctive member of the genus that was previously placed in its own section (Beardsley et al. 2004). Until recently, Mojave monkeyflower was included in the figwort family (*Scrophulariaceae*), but it is now placed in the lopseed family (*Phrymaceae*) (Beardsley and Olmstead 2002; Jepson Flora Project 2011). There are also current studies that provide evidence that the genus *Mimulus* should be fragmented into several new genera, so more nomenclatural changes can be expected in the near future for this taxon.

Mojave monkeyflower is an annual plant approximately 2 to 10 centimeters (0.8 to 3.9 inches) in size. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

This species occurs in the Mojave Desert in west-central San Bernardino County (Jepson Flora Project 2011). The populations with greatest known densities occur south of Daggett and Barstow (MacKay 2006). However, the majority of the historical occurrences

¹ **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly endangered in California.

in the Barstow area have either been extirpated or impacted (CNPS 2011). The elevation range of this species extends from 600 to 1,200 meters (1,969 to 3,937 feet) (CNPS 2011) (Figure SP-P06). There are a total of 56 California Natural Diversity Database (CNDDB) occurrences for Mojave monkeyflower at 121 localities, all of which occur in the Plan Area.

Distribution and Occurrences within the Plan Area

Historical

Eleven localities have not been observed since 1990. Of these, one site at Kane Springs (Element occurrence 6) was visited more recently (in 2011) and no plants were found so it is uncertain whether any plants occur here. However, the Kane Springs resurvey in 2011 with negative results does not mean the plants are not in the vicinity (MacKay, pers. comm. 2012). One occurrence along Camp Road is not dated and no plants were found at this site in 1986 or in 1998. Moore (pers. comm. 2012) stated that areas off of Camp Rock Road on the smaller BLM roads represent important Mojave monkeyflower habitat because they have very low levels of disturbance in comparison to those on Camp Rock Road. Another historical occurrence is the type locality in Calico and is likely extirpated (CDFW 2013). These records extend from the area around Barstow southeast to the area around the Newberry Mountains, and one occurrence much farther south near Old Woman Springs (Figure SP-P06) (CDFW 2013).

Recent

Of the 121 total CNDDB localities in the Plan Area, 110 have been recorded in the CNDDB since 1990 and are presumed extant. One of the major populations of Mojave monkeyflower recorded in the CNDDB since 1990 that is presumed extant is located southeast of Barstow to Ord Mountain. A second concentration of occurrences is located northeast of Adelanto and extends to Helendale. There is an isolated occurrence just south of the Black Mountains summit (Figure SP-P06). However, if the Stoddard Open Off-Highway Vehicle (OHV) area were surveyed there is a high likelihood that Mojave monkeyflower would be documented, providing a continuum of distribution between the two major areas (MacKay, pers. comm. 2012). The disjunct distributions are the Kane Springs collection east of Rodman (Element occurrence 6) and

the Old Woman Springs collection; both areas still need field work (MacKay, pers. comm. 2012).

According to CNDDDB records (CDFW2013), of the 47 current occurrences at 110 localities, the vast majority are on lands managed by the Bureau of Land Management (BLM), and the remaining portion are on lands that are privately owned or whose ownership is unknown (CDFW 2013). However, 14 of the 19 occurrences turned in by B. West (BLM employee at the time, 1992) included information that the BLM-owned lands were under consideration for disposal, and BLM subsequently disposed of the land containing four of those occurrences (CDFW 2013; MacKay, pers. comm. 2012). Also, there is a very high probability that the remaining Brisbane Valley is occupied by Mojave monkeyflower (MacKay, pers. comm. 2012).

Natural History

Habitat Requirements

This species occurs in Mojavean desert scrub, specifically creosote bush scrub (MacKay 2006; CNPS 2011). Mojave monkeyflower is associated with the following species or genera, among others: creosote bush (*Larrea tridentata*), desert senna (*Senna armata*), cheese bush (*Ambrosia salsola*), ratany (*Krameria erecta* and *K. bicolor*), chollas (*Cylindropuntia* spp.), burro bush (*Ambrosia dumosa*), prairie-clovers (*Psoralea* spp.), Bigelow's monkeyflower (*Mimulus bigelovii*), desert bells (*Phacelia campanularia*), desert fivespot (*Eremalche rotundifolia*), spiny hopsage (*Grayia spinosa*), and desert trumpet (*Eriogonum inflatum* var. *inflatum*) (MacKay 2006; CDFW 2013).

Mojave monkeyflower commonly occurs in areas that are not subject to regular water flow (MacKay 2006). These areas include the gravelly banks of desert washes with granitic soils and rocky slopes above washes, as well as the sandy openings of creosote bush scrub (MacKay 2006).

Table 1. Habitat Associations for Mojave Monkeyflower

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mojavean desert scrub, Creosote bush scrub	Primary habitat	Granitic soils, 1,968–3,937 feet	MacKay 2006; CNPS 2011; Jepson Flora Project 2011

Reproduction

Germination is probably dependent upon the amount of precipitation, as population sizes can vary substantially from year to year (MacKay 2006).

Most members of the lopseed family are insect pollinated (Beardsley and Olmstead 2002); and given the showy flowers, Mojave monkeyflower pollinators are probably Hymenoptera (bees, wasps, ants, and sawflies) or Lepidoptera (butterflies and moths). MacKay (2006) hypothesized that the white margin of the corolla reflects ultraviolet light, and the maroon veins extending into this margin act as nectar guides to facilitate pollination.

Small seeds and an annual habit suggest that dispersal of Mojave monkeyflower is mostly abiotic (MacKay 2006; NatureServe 2010). For populations located on rocky slopes above washes, it is probable that gravity carries seeds down into the washes and intermittent water flow may carry seeds further down washes. Although biotic vectors of seed transport are unknown, granivorous ants or rodents may transport seeds over short distances and birds may transport seeds longer distances (MacKay 2006).

Ecological Relationships

Although suitable habitat for this species appears to be fairly abundant, it is quite restricted geographically. Population sizes fluctuate substantially from year to year, probably in response to the amount and timing of precipitation; as an annual, germination and establishment are dependent on the timing and amount of spring rains (MacKay 2006; NatureServe 2010). Unknown unusual germination and establishment requirements may account for the considerable variability in population sizes from year to year (MacKay 2006).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)

State: S2, Imperiled (CDFG 2012b)

Population trends for Mojave monkeyflower are unknown at present, but a multi-year population-level study is underway by BMP Ecosciences (Moore et al.) and expected to be completed by 2015. One CNDDDB locality has been possibly extirpated, and the status of 11 of the 121 total CNDDDB localities of Mojave monkeyflower in the Plan Area have not been updated since 1990 (CDFW 2013; MacKay 2006).

Threats and Environmental Stressors

Threats to Mojave monkeyflower include development, mining, non-native plants, solar and wind energy projects, grazing, vehicles, and road development (CNPS 2011; NatureServe 2010; MacKay 2006). Additional potential threats include pipeline installation and quarries and test pits adjacent to populations (MacKay 2006). Mojave monkeyflower is also under threat by the potential for the BLM to convert land occupied by this species to private lands, which could then be developed (MacKay 2006; CDFW 2013). The area under consideration for disposal or land exchange is located between Barstow and Victorville (CDFW 2013).

Because population sizes fluctuate considerably annually in response to environmental conditions, Mojave monkeyflower is susceptible to depletion of the seed bank after a series of drought years. In addition, small population sizes increase the risk of inbreeding, which may result in reduced seed set or reduced seed viability (MacKay 2006).

Conservation and Management Activities

The West Mojave Plan designated Mojave monkeyflower conservation areas in the Plan Area as land managed by BLM (BLM 2005). The Brisbane Valley Mojave Monkeyflower Conservation Area is 10,448 acres and the Daggett Ridge Mojave Monkeyflower Conservation Area is 25,351 acres (BLM 2006).

Data Characterization

In general, data availability for the Mojave monkeyflower is poor. The pollination ecology of Mojave monkeyflower is unknown (MacKay 2006). This species may have some unusual germination and establishment requirements that are unknown (MacKay 2006). Mojave monkeyflower is also absent from much apparently suitable habitat and remains relatively restricted geographically (MacKay 2006).

The status of many of the recorded populations of Mojave monkeyflower is unknown. Several occurrences documented in the CNDDDB may be extirpated but still presumed extant in the database (MacKay 2006). In addition, location data may be inaccurate, especially for older records labeled Barstow; these collections may actually be from the vicinity of Barstow, and not from what is now the town of Barstow (MacKay 2006).

Management and Monitoring Considerations

Protection of the areas where Mojave monkeyflower is known to occur is important to maintain viable populations of the species. The species would likely benefit from the elimination of off-road vehicle use and livestock grazing in occupied areas south of Barstow and Daggett, as well as maintenance of BLM management of lands between the Mojave River and Interstate 15 between Victorville and Barstow. Management and monitoring are complicated by the year-to-year fluctuations in population size in response to rainfall. A very important consideration is to fully understand where populations occur. Vast and thorough surveys should be conducted during the appropriate flowering season in good rainfall years (MacKay, pers. comm. 2012). Confirmation of site occupancy in suitable habitat should be conducted over multiple years before concluding absence. Moore et al. (in prep) found that novel occurrence discoveries in modeled suitable habitat were strongly predicted by the proximity to recent occurrences.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mojave monkeyflower, using available spatial information and occurrence

information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 176,190 acres of modeled suitable habitat for Mojave monkeyflower in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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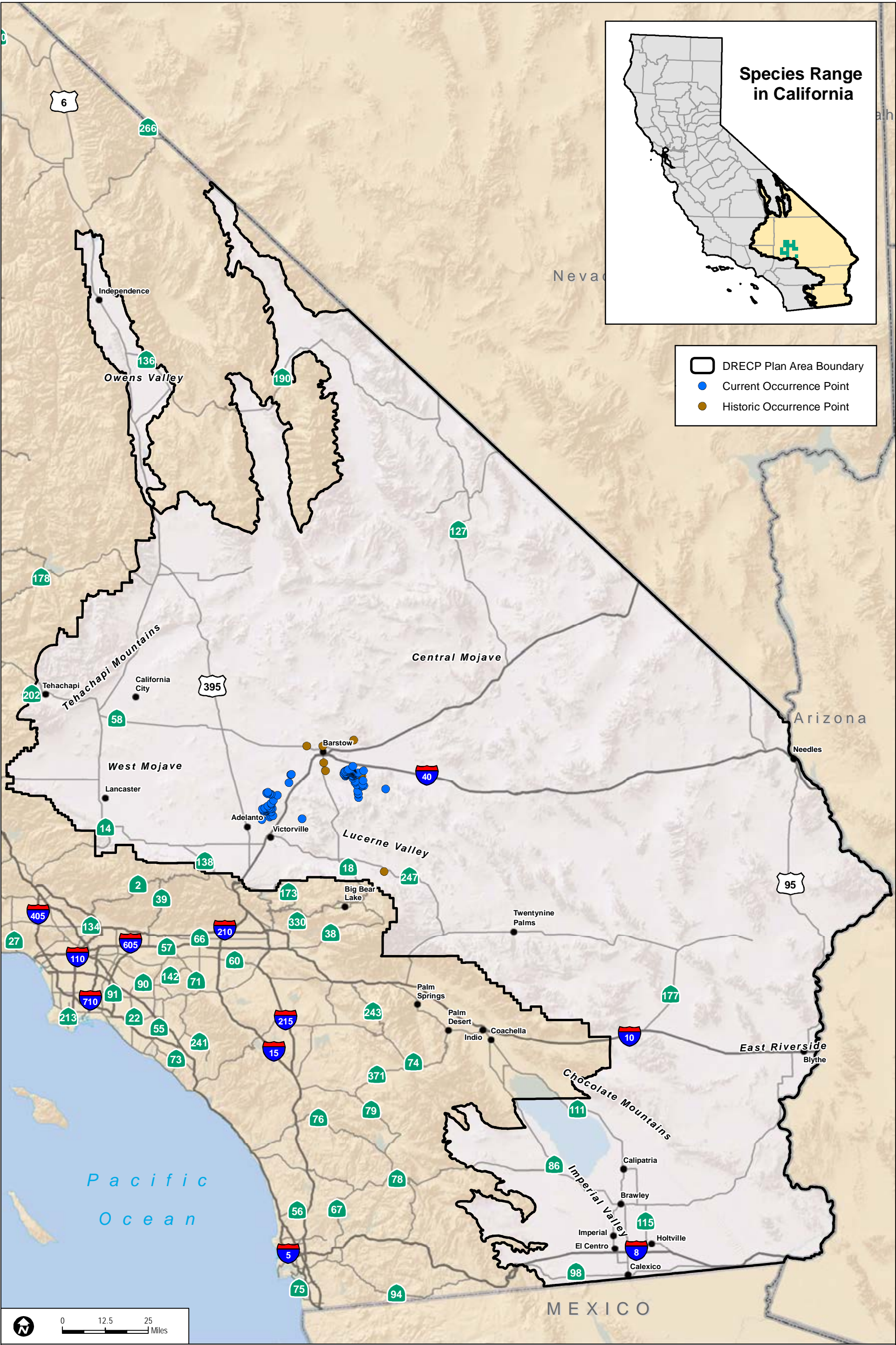
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P06
Mojave Monkeyflower Occurrences in the Plan Area

Mojave Tarplant (*Deinandra mohavensis*)

Legal Status

State: Endangered; S2S3¹

California Rare Plant

Rank: 1B.3²

Federal: Bureau of Land Management Sensitive; U.S. Forest Service Region 5 Sensitive Plant Species

Critical Habitat: N/A

Recovery Planning: N/A



Photo courtesy of Heath McAllister.

Taxonomy

Mojave tarplant is in the sunflower family (Asteraceae) (Jepson Flora Project 2011). Mojave tarplant was originally described by D.D. Keck (1935) as *Hemizonia mohavensis* and was reclassified as *Deinandra mohavensis* in 1999 (Baldwin 1999). The taxonomic revision was intended to more accurately reflect phylogenetic relationships within Madiinae (a subtribe within Asteraceae) (Baldwin 1999). The plant was thought to be extinct but was rediscovered by A. Sanders in 1994 in the San Jacinto Mountains, in Riverside County (Sanders et al. 1997).

Mojave tarplant is an annual plant approximately 10 to 100 centimeters (3.9 to 39 inches) in height. A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011).

Distribution

General

There are a total of 75 occurrences in the California Natural Diversity Database (CNDDB) at 124 localities (CDFW 2013a). Mojave tarplant is known in Kern, Riverside, Inyo, and San Diego counties (believed extirpated from San Bernardino County) (CDFW 2013a) (Figure SP-P07). This species occurs at elevations of 460–1,600 meters (1,509–

¹ **S2S3:** the rank is somewhere between S2, Imperiled and S3, Vulnerable.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.3:** Not very endangered in California.

5,250 feet) (CNPS 2011; Jepson Flora Project 2011). The distribution is discontinuous and possibly relictual.

Distribution and Occurrences within the Plan Area

Historical

This species was not known to occur in the Plan Area prior to 1990 (CDFW 2013a; Figure SP-P07).

Recent

Within the Plan Area, Mojave tarplant is known from the desert slope of the southern Sierra Nevada Mountains in Kern County (Sanders 2006a). There are 10 occurrences at 13 localities in the Plan Area, all within Kern and Inyo counties. The majority of localities are located west of Highway 14 and east of the Sequoia National Forest; north of Interstate 40; near Cutterbank Spring; in Jawbone Canyon; near Short Canyon; in lower Esperanza Canyon; in lower Water Canyon; and in the vicinity of Cross Mountain (CDFW 2013a) (Figure SP-P07). Mojave tarplant may also occur at Red Rock Canyon in Red Rock Canyon State Park in Kern County (Faull, pers. comm. 1998, cited in Sanders 2006a).

Natural History

Habitat Requirements

Mojave tarplant occurs in open moist sites in arid regions near the margins of the desert, within chaparral, coastal scrub, desert scrub, riparian scrub, and woodland (CNPS 2011; Sanders 2006a; Jepson Flora Project 2011). Plants are typically observed in seeps and along grassy swales and intermittent creeks. The most suitable habitat occurs in mountainous areas within microhabitats of low gradient streams and on gentle slopes with few shrubs and trees. This species is associated with clay or silty soils that are saturated with water early in the year. Mojave tarplant prefers areas that are dry at the surface but which have a substantial water source at depth through summer. Dwarfed plants occasionally are found in drier sites near occupied moist areas (Sanders et al. 1997). This cycle of early saturation with later desiccation may

reduce competition from other plant species; dryness during drought years may further reduce competition (Sanders 2006a).

At the type locality, Mojave tarplant was known to occur along a sandy intermittent creek; however, this habitat is now believed to be atypical and not suitable to maintain a permanent population. Sanders et al. (1997) note that some occurrences of Mojave tarplant are associated with sand where the sand is adjacent to more typical habitat.

Table 1. Habitat Associations for Mojave Tarplant

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mesic openings in chaparral, desert and coastal scrub, woodland, and riparian scrub	Primary	Clay or silty soils (sometimes sand); seasonally (winter and spring) saturated with water; 460–1,600 meters (1,509–5,250 feet)	CNPS 2011; Sanders et al. 1997; Sanders 2006a; Jepson Flora Project 2011

Reproduction

Mojave tarplant is an annual plant that blooms from June through January (CNPS 2011). Flowering peaks between August and October. Once flowering has begun, it continues until the plants begin to senesce. Fruit maturity and dispersal are continuous as well. Unlike most former *Hemizonia* species, including the segregated *Deinandra*, Mojave tarplant is self-compatible (Baldwin pers. comm. 1998, cited in Sanders 2006a); the only other self-compatible member of *Deinandra* is Red Rock tarplant (Tanowitz 1982). Pollination studies have not been conducted for Mojave tarplant; however, Faull (1987) observed small beetles and honey bees visiting Red Rock tarplant flowers, a closely related species.

Mojave tarplant blooms from June through January (CNPS 2011). Flowering peaks between August and October. Once flowering has begun, it continues until the plants begin to senesce. Fruit maturity and dispersal are continuous as well. Seed dispersal vectors have not been reported for this species; however, the seeds are relatively heavy and may just fall to the ground around the source plant. The seeds are

not armed with any obvious mechanisms, such as hooks or wings, for long-distance dispersal (Sanders 2006a). Bruce Baldwin (pers. comm., cited in Sanders 2006b) reports that ray achenes of *Hemizonia* (including the segregated *Deinandra*) maintain some degree of dormancy while the disk achenes freely germinate.

Mojave tarplant is known to reproduce easily in cultivation (B. Baldwin, pers. comm. 1998, cited in Sanders 2006a) and has been known to colonize disturbed areas in a botanical garden (S. Boyd, pers. comm. 1998, cited in Sanders 2006a).

Ecological Relationships

As described in Habitat Requirements, Mojave tarplant is associated with seasonally saturated clay or silty soils on gentle slopes or low gradient streams, with few shrubs and trees. These saturated areas are typically dry at the surface but provide a substantial water source at depth through summer (Sanders et al. 1997). This species has a discontinuous and possibly relictual distribution (Sanders 2006a), and little is known of its life history and ecological relationships. Although pollination studies have not been conducted for Mojave tarplant, Faull (1987) has observed small beetles and honey bees visiting Red Rock tarplant flowers, a closely related species. Seed dispersal vectors have not been reported for this species; however, the seeds are relatively heavy and may just fall to the ground around the source plant. The seeds are not armed with any obvious mechanisms, such as hooks or wings, for long-distance dispersal (Sanders 2006a). Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). Within the Plan Area, intense cattle grazing and trampling may be the most significant threats.

Population Status and Trends

Global: G2G3, Imperiled/Vulnerable (NatureServe 2011, Conservation Status last reviewed XXXX)

State: S2S3, Imperiled/Vulnerable (CDFW 2013b)

Because this species was only recently rediscovered (in 1994) there is little information available on population trends. Of the 13 occurrences in the Plan Area, four are on BLM lands, two are on private land, and

ownership is unknown for two of the occurrences. The occurrence on private land near Cutterbank Spring numbered 14 individuals in 2003. Approximately 15,000 plants were observed at the other occurrence on private land located at the south end of Kelso Valley in 2010. Many more plants were observed in 2011, including an additional 1,500 plants in the northeastern portion of the occurrence (CDFW 2013a). Of the two occurrences for which ownership is unknown, one numbered in the thousands in 1998 and the other numbered 109 individuals in 2003. Of the four occurrences on BLM land, one numbered 50,000 in 2003 (with 30 rosettes observed very early in the year in 2004), one numbered in the several hundreds in 2008, and one numbered 5,000 in 1998 (and was locally common in 2001 and numbered 3,000 in 2003). Approximately 50,000 plants were observed in 2003 at the occurrence at Cutterbank Spring on BLM lands; 30 plants were observed in 2004 in their rosette form in an early season survey, and plants were “abundant around the springs and in the surrounding drainage channels” in 2010 (CDFW 2013a). Overall, there are 69 occurrences in Kern, Riverside, and San Diego counties (CDFW 2013a) and most of these appear to have number of individuals estimated only once, making it difficult to discern a population trend.

Threats and Environmental Stressors

Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). The type locality was modified by construction of the Mojave River Forks Dam. Within the Plan Area, cattle grazing occurs at some of the Mojave tarplant occupied areas, and in some areas is locally intense and may pose a threat. However, the sticky plants of the genus *Deinandra* (also called “tarweeds”) may not be palatable to cattle, so grazing may not be a major threat and trampling by cattle around limited watering sources in dry areas may be a greater threat (Sanders 2006a).

Conservation and Management Activities

Four of the occurrences are known from BLM land, two are on private land, and ownership is unknown for two of the occurrences (CDFW 2013a). No current conservation or management activities have been identified for Mojave tarplant.

Data Characterization

The general distribution of Mojave tarplant is discontinuous and patchy. Sanders (2006a) recommends that additional surveys be conducted in the southern Sierra Nevadas and along the north foot of the Transverse Range, particularly the San Gabriel Mountains. Within the Plan Area, four of the occurrences are known from BLM lands, two are on private land, and ownership is unknown for two of the occurrences. Many of the known occurrences outside the Plan Area occur within the San Bernardino and Cleveland National Forests and therefore receive some protection (Sanders 2006a). The species is known to be self-compatible (B. Baldwin, pers. comm. 1998, cited in Sanders 2006a) and a related species (Red Rock tarplant) is known to be insect-pollinated (Faull 1987). Little is known regarding the species' seed dispersal and recruitment.

Management and Monitoring Considerations

Because the global distribution of Mojave tarplant is discontinuous and patchy, Sanders (2006a) recommends that additional surveys be conducted in the southern Sierra Nevadas and along the north foot of the Transverse Range, particularly the San Gabriel Mountains. Additional surveys may identify new occurrences.

Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). Measures to control these threats should be considered.

Species Modeled Habitat Distribution

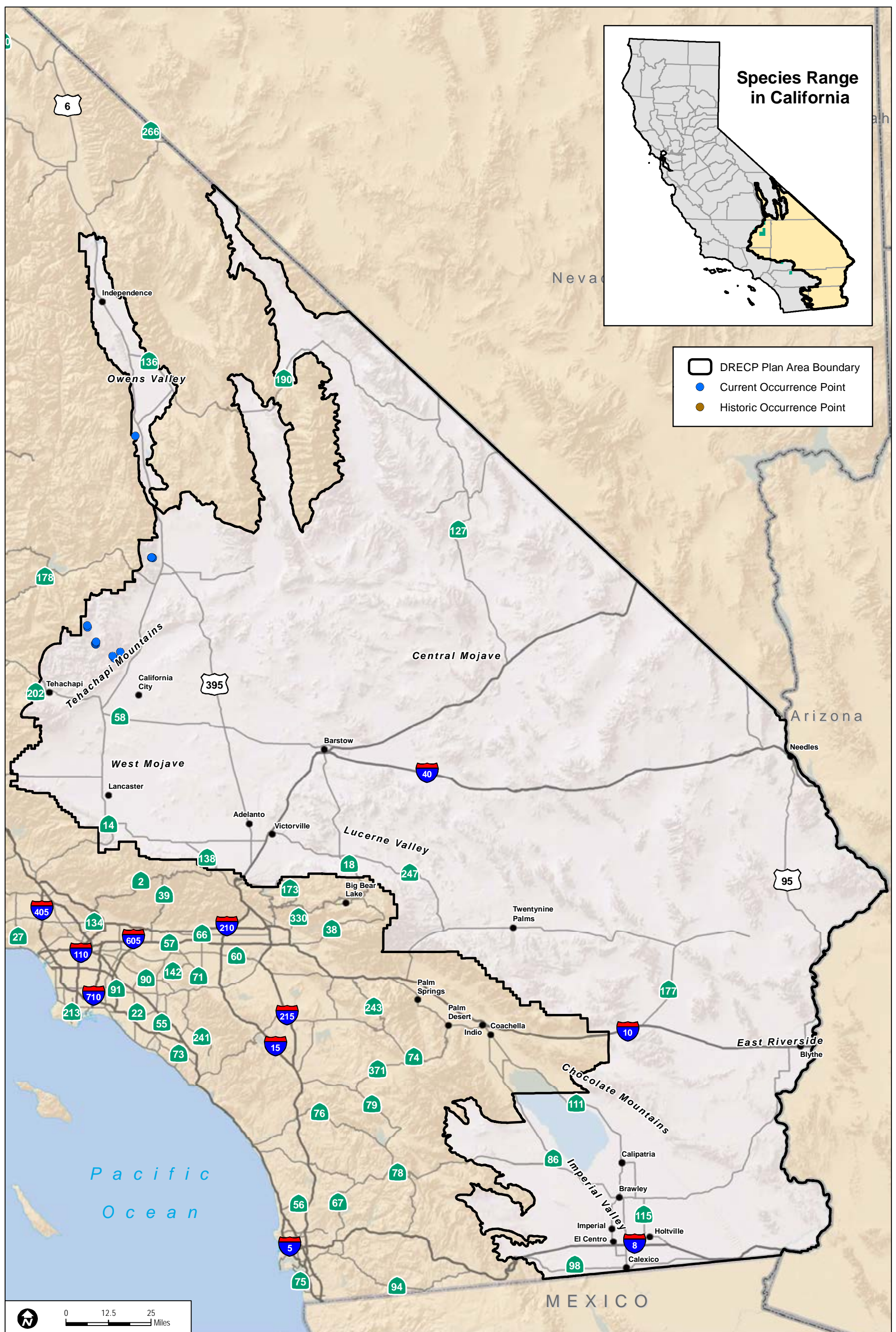
This section provides the results of habitat modeling for Mojave tarplant, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 270,463 acres of modeled suitable habitat for Mojave tarplant in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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Owens Valley Checkerbloom

Sidalcea covillei

Legal Status

State: Endangered; S3¹

California Rare Plant

Rank: 1B.1²

Federal: Bureau of Land Management Sensitive

Critical Habitat: None

Recovery Planning: *Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties, California* (USFWS 2000)

Notes: Considered for federal listing (proposed as a candidate species) in 1985, it was removed from the candidate list in 1996 because the U.S. Fish and Wildlife Service (USFWS) determined that the species was more abundant or widespread than was previously thought, or the species was not subject to any identifiable threat.



Photo courtesy of Larry Blakely.

Taxonomy

Owens Valley checkerbloom (*Sidalcea covillei*) was originally described by E. Greene in 1914 and the taxonomic status of Owens Valley checkerbloom has not changed since it was first described.

Owens Valley checkerbloom is a perennial herb with stems approximately 2 to 6 decimeters (7.9 to 24 inches) in length. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

Owens Valley checkerbloom is endemic to the southern Owens Valley in Inyo County, California (CNPS 2011; BLM 2011b). It grows only in alkali meadow and spring communities scattered along about 125 kilometers (77.7 miles) of the Owens River drainage (Halford 1994). The California

¹ **S3:** Vulnerable.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously endangered in California.

Natural Diversity Database (CNDDDB) includes 42 occurrences of Owens Valley checkerbloom at 35 localities; 21 of these occurrences are in the Plan Area at 30 localities.

Distribution and Occurrences within the Plan Area

Historical

Owens Valley checkerbloom was first collected in 1891 in an extensive alkali meadow known as Haiwee Meadows, Inyo County, and was not collected again until 1952, when it was found north of Lone Pine in Inyo County. The species was extirpated from its type locality when the Haiwee Reservoir was formed, and by 1978, local botanist Mary DeDecker considered it to be on the brink of extinction (DeDecker 1978). Within the Plan Area, 5 of the 30 known localities are considered historical (i.e., pre-1990) and have not been recently observed. These populations are known to be either extirpated, possibly extirpated, or are presumed to be extant (CDFW 2013a).

Recent

The CNDDDB includes 25 recent localities (i.e., since 1990) of Owens Valley checkerbloom in the Plan Area. All of these localities occur on lands owned by the LADWP (CDFW 2013a). All of the localities are generally along Highway 395 from the meadow above Tinemaha Creek south to the area 1 mile north of Olancho (Figure SP-P08) (CDFW 2013a).

Natural History

Habitat Associations

Owens Valley checkerbloom grows in moist alkaline meadows and seeps at elevations of 3,580 to 4,650 feet (see Table 1; CNPS 2011; CDFW 2013a). Almost all occurrences grow in fine, sandy loam with alkaline crusts, but one occurrence is known to grow in stony, calcareous soil (CDFW 2013a).

Associated native grasses and herbs include saltgrass (*Distichlis spicata*), alkali sacaton (*Sporobolus airoides*), basin wildrye (*Elymus cinereus*), Baltic rush (*Juncus balticus*), and clustered field sedge

(*Carex praegracilis*). Associated shrubs at some sites include basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*). The endemic Inyo County star-tulip (*Calochortus excavatus*) co-occurs with Owens Valley checkerbloom at some sites (Halford 1994).

Table 1. Habitat Associations for Owens Valley Checkerbloom

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Meadows and seeps	Primary	Alkaline soils; 3,580–4,650 feet elevation	CDFW 2013a; CNPS 2011

Reproduction

Owens Valley checkerbloom flowers from April through June (BLM 2011b; CNPS 2011). The pink-lavender flowers are showy and Owens Valley checkerbloom is probably an outcrossing species that is pollinated by insects. Bees are major pollinators in other related *Sidalcea* species (summarized in Leong 2006). The breeding system of Owens Valley checkerbloom is not known, but research on related *Sidalcea* species has found that several species are gynodioecious, meaning that some plants bear hermaphrodite flowers and other plants bear female-only flowers (Leong 2006). Low seed germination rates in Owens Valley checkerbloom have been reported in one study, ranging from 1.6% to 12.5% (Halford 1994). The Halford (1994) study suggested that seed weight may influence germination rates, with heavier seeds producing higher germination rates; plants may produce larger seeds in favorable years. Plant reproduction was reduced by high rates of rabbit and rodent herbivory on study sites (Halford 1994). This study identified that germination rates for Owens Valley checkerbloom may be enhanced through minor treatments such as leaching or cold stratification and mild giberellic acid treatments.

Ecological Relationships

Owens Valley checkerbloom occurs solely in mesic high-elevation alkaline meadows habitats in the Owens Valley River drainage. This species is highly restricted to a specialized habitat with very limited distribution.

The Owens Valley checkerbloom may be highly sensitive to drought conditions, although DeDecker (1978) suggested that the fleshy roots might help it survive normal drought cycles; individuals observed during the low rainfall years of 1993 and 1994 yielded low weight seeds with low viability (Halford 1994). In addition, local drought conditions may result in more browsing by rabbits and rodents, which in turn can reduce seed set and reproduction of the species (Halford 1994).

Population Status and Trends

Global: G3, Vulnerable (NatureServe 2011, Conservation Status last reviewed 2006)

State: S3, Vulnerable (CDFW 2013b)

The very restricted range and few population occurrences of Owens Valley checkerbloom make it vulnerable to declines from a variety of threats, including natural and anthropogenic sources described under Threats and Environmental Stressors. Due to the lack of long-term surveys, censuses, and/or monitoring studies, population trends of the species are unknown.

Threats and Environmental Stressors

The diversion of the Owens River and cattle grazing were the main causes of this species' decline to near extinction (DeDecker 1978). Halford (1994) reported that low annual precipitation, improper timing and intensity of cattle grazing, increased competition from rhizomatous grass species and upland shrubs, and diversions or depletions of naturally occurring water sources are all threats to the species. Lowering of the local water table by pumping and drainage for water diversion, and the resultant invasion of non-native plants, or heavy grazing and associated meadow succession may be a major threat (Hill 1993). Elmore et al. (2006), for example, reported that alkali meadow vegetation in the Owens Valley is groundwater-dependent and plant cover at groundwater-depleted sites is only weakly correlated with precipitation. Grazing, mostly by cattle, is the most frequently mentioned threat in CNDDDB records (CDFW 2013a). Noxious weeds such as Russian olive (*Elaeagnus angustifolia*) and knapweed (*Centaurea* spp.) occur at a couple of occurrences, and invasion of rubber rabbitbrush (*Ericameria nauseosa*) may result from lowering of the water table.

Conservation and Management Activities

According to the CNDDDB, Owens Valley checkerbloom is restricted to approximately 42 occurrences in Inyo County, of which 22 are in the Plan Area (CDFW 2013a). A cooperative project was initiated in 1994 by the BLM, the California Department of Fish and Wildlife, and The Nature Conservancy to test the long-term survivorship of reintroduced Owens Valley checkerbloom. Seeds were collected from several populations, subjected to several experimental treatments, and sown at a local nursery, and the seedlings (136 in total) were reintroduced back into sites from which the seed was collected. All plants had a minimum of a 30-centimeter (12-inch) root system when planted in October 1994, and survivorships of 50% and 85% were reported from the two sites afterwards (BLM 1994). The success of this project demonstrates that the species can be successfully propagated and transplanted, allowing some flexibility in the response of management activities to suitable habitat areas disturbed by grazing or other surface disturbing threats. However, as noted above under Threats and Environmental Stressors, groundwater management is likely a key consideration for successfully conserving and managing this species.

In 2011, the Bishop Paiute received a \$200,000 grant from the USFWS to reintroduce, sustain, and nurture populations of several rare plants, including Owens Valley checkerbloom on tribal lands in the Owens Valley (USFWS 2011).

Data Characterization

An information gap extends from the mid-1990s through today. Long-term surveys, censuses, and/or monitoring studies have not been conducted on Owens Valley Checkerbloom since the mid-1990s.

Management and Monitoring Considerations

As identified under Threats and Environmental Stressors, cattle grazing, groundwater depletion, and the associated invasion by competing species are probably the main threats to Owens Valley checkerbloom. Further study regarding the response of Owens Valley checkerbloom to these factors is needed (Halford 1994). There is no specific information available on pollinators or breeding system.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens Valley checkerbloom, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 147,869 acres of modeled suitable habitat for Owens Valley checkerbloom in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

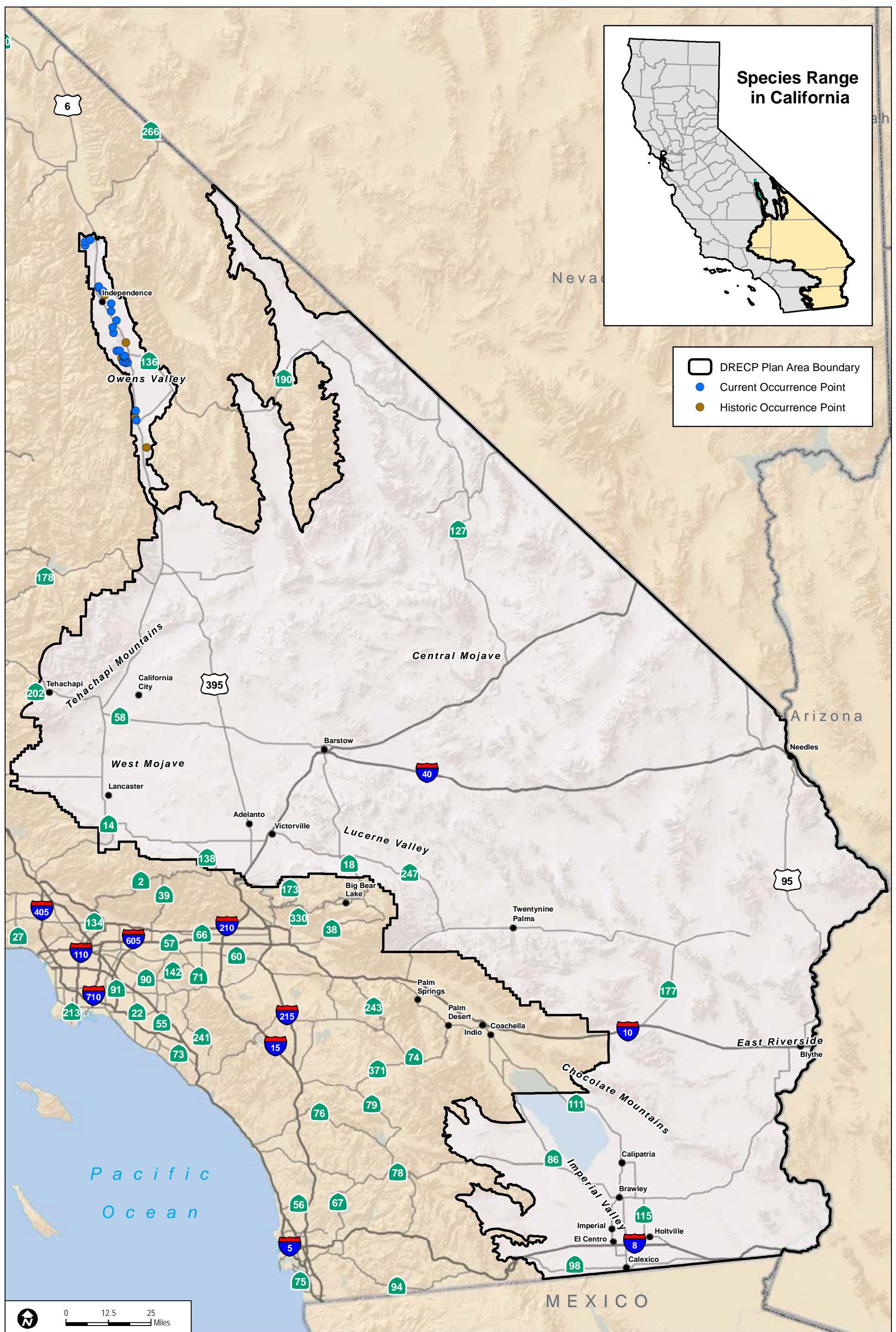
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Parish's Daisy (*Erigeron parishii*)

Legal Status

State: S2S3¹

CNPS: Rare Plant Rank 1B.1²

Federal: Threatened

Critical Habitat: Originally designated on December 12, 2002 (67 FR 78570–78610).

Recovery Planning: *San Bernardino Mountains Carbonate Plants Draft Recovery Plan* (USFWS 1997)

Notes: No status changes predicted by U.S. Fish and Wildlife Service (USFWS) in 2010 (75 FR 28636–28642)



Photo courtesy of Duncan S. Bell.

Taxonomy

Parish's daisy (*Erigeron parishii*) was named by Asa Gray in 1884 in his *Synoptical Flora of North America* and has remained stable with no changes since. Parish's daisy is in the sunflower family (Asteraceae) (IPNI 2011). It is an herbaceous perennial subshrub approximately 7 to 30 centimeters (3 to 12 inches) in height from its taproot. A full physical description of the species can be found in *Jepson eFlora* (Jepson Flora Project 2012).

Distribution

General

Parish's daisy is endemic to Southern California, restricted to dry, calcareous (mostly limestone) slopes of the San Bernardino Mountains, with a few collections from granitic areas at the east end of the San Bernardino Mountains and in the Little San Bernardino Mountains (Neel 2000; Sanders 2006). Parish's daisy occurs at elevations between 3,700 and 6,600 feet, most often in washes and canyon bottoms, but sometimes on alluvial benches or steep rocky

¹ **S2:** Imperiled.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously endangered in California.

mountainsides (Mistretta and White 2001). It is estimated that 1,029 acres are occupied Parish's daisy habitat (USFWS 2009).

Distribution and Occurrences within the Plan Area

Historical

Parish's daisy was first described by Asa Gray in 1884 from specimens collected by S.B. Parish at Cushenbury Springs in May 1881 (Abrams and Ferris 1960; Krantz 1979). It was reported to be "abundant on stony hillsides at Cushenberry Springs" by Hall (1907), although it is unclear whether Hall was referring to Parish's collections of the species (Sanders 2006). Within the Plan Area, the California Natural Diversity Database (CNDDB) includes two historical occurrences that were documented in 1988 and two historical occurrences for which status is unknown (Figure SP-P09). However, each of these occurrences is presumed to be extant.

Recent

Within the Plan Area, the CNDDB includes 40 recent occurrences (i.e., post-1990) of Parish's daisy and all are regarded as extant (CDFW 2013a) (Figure SP-P09). The populations occur primarily on U.S. Forest Service (USFS) and BLM lands, but two of the populations on USFS and BLM lands also extend onto private lands within the Plan Area. Two populations occur within the Joshua Tree National Park and another is located on the University of California Natural Reserve System Burns Pinion Ridge Reserve (CDFW 2013a).

In 2009 the USFWS determined that the range and distribution of this species was essentially the same as it was at the time of listing (1994).

Natural History

Habitat Requirements

Parish's daisy occurs in Mojavean desert scrub and pinyon and juniper woodlands (CNPS 2011) and is largely restricted to loose, carbonate alluvium, although it is occasionally found on other rock types (Sanders 2006) (Table 1). Populations of Parish's daisy are most commonly found along washes on canyon bottoms or on loose alluvial

deposits on adjacent benches, but they are also occasionally found on steep rocky slopes (Sanders 2006). Based on this species' occurrence on noncarbonate granitic soils, it is possible that the apparent carbonate preference is due to reduced competition from other plants, although reports of this species on noncarbonate soils are few (Sanders 2006). It has also been observed at sites where soils have been found to be strongly alkaline, implying that the noncarbonate granitic soils may have been influenced in their soil chemistry by adjacent carbonate slopes (Sanders 2006).

Specific plant species associated with Parish's daisy have not been described in the literature, but dominant species within pinyon and juniper woodland where Parish's daisy is typically found include singleleaf pinyon pine (*Pinus monophylla*), Utah juniper (*Juniperus osteosperma*), and more rarely California juniper (*Juniperus californica*) and western juniper (*Juniperus occidentalis*). Understory species within pinyon and juniper woodland are more variable, but may include mountain-mahogany (*Cercocarpus ledifolius*), Mormon tea (*Ephedra viridis*), Mojave yucca (*Yucca schidigera*), Joshua tree (*Yucca brevifolia*), and encelia (*Encelia* sp.).

Parish's daisy co-occurs with another carbonate endemic, Cushenbury oxytheca (*Acanthoscyphus parishii* var. *goodmaniana*). Its presence, however, appears to be negatively related to at least two other carbonate soils species—Cushenbury milk-vetch (*Astragalus albens*) and Cushenbury buckwheat (*Eriogonum ovalifolium* var. *vineum*)—which tend to occur on more stable slopes.

Table1. Habitat Associations for Parish's Daisy

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Pinyon-juniper woodland, Joshua tree woodland, Mojavean desert scrub, Jeffrey pine-western juniper woodland	Primary habitat	Carbonate soils (limestone), 3,000 to 6,600 feet	Sanders 2006; USFWS 2009

Reproduction

Parish's daisy is a long-lived perennial (Mistretta and White 2001) that flowers from May through August (CNPS 2011), peaking mid-May to mid-June (Sanders 2006). Based on the conspicuous flowers, pollinators are probably insects and would include bees, butterflies, and other known pollinators of similar and related species (Sanders 2006). Parish's daisy produces plumed achenes adapted for wind dispersal (Mistretta and White 2001) and does not appear to have a seed dormancy mechanism (Mistretta 1994). Based on observations of seedlings at several sites (Krantz 1979), reproduction is probably primarily by seed rather than vegetatively by rhizomes or stolons. A recent study by Neel and Ellstrand (2001) found no evidence of vegetative reproduction, concluding that the species probably primarily reproduces sexually through outcrossing.

Recent research on allozyme diversity showed that genetic diversity was high (compared to many narrowly endemic plant taxa) and populations were only moderately differentiated, suggesting that gene flow among populations is still high and any recent fragmentation has not yet affected genetic diversity. Maintaining the existing large population sizes is an important component in maintaining gene flow among populations (Neel and Ellstrand 2001).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)

State: S2, Imperiled (CDFW 2013b)

The current population status of Parish's daisy is unclear and there is a discrepancy in total reported occurrences of the species. According to the final listing rule in 1994, Parish's daisy was known from fewer than 25 occurrences with a total estimated population size of 16,000 individuals, but at that time, the San Bernardino National Forest had mapped 87 site-specific occurrences (USFWS 2009). USFWS (2009) notes that what constitutes an occurrence has been subjectively defined over various surveys, making it difficult to specify status or change in status of Parish's daisy since it was listed. In addition, there has been an increase in survey efforts for this species since listing that has resulted in an increase in the number of occurrences detected.

Sanders (2006) characterizes Parish's daisy as one of the more common carbonate endemics of the San Bernardino Mountains. Nonetheless, there have not been any systematic population studies conducted over time to document population trends.

Threats and Environmental Stressors

The main threat to Parish's daisy is limestone mining because this species is mostly restricted to carbonate deposits (USFWS 2009). Besides direct impacts, dust and artificial lighting can affect the species through dust impacts on soil chemistry and lighting availability for seeds and the impacts of artificial lighting on growing conditions (USFWS 2009). Sanders (2006) notes that after moistening, the mining dust appears to harden into a cement-like coating. Additional threats listed by USFWS and CNPS include energy development projects, off-highway vehicles, fuel-wood collection, fire suppression activities, camping, target shooting, road construction, and residential developments, but these threats are relatively low compared to mining (USFWS 2009; CNPS 2011).

The specific potential effects of climate change on Parish's daisy are unknown, but if climate change caused a shift to higher elevations due to warmer and drier conditions, as has occurred with other plant species on the Santa Rosa Mountains of Southern California (Kelley and Goulden 2008), this endemic species could be concentrated in a smaller area and more vulnerable to extinction (USFWS 2009).

Conservation and Management Activities

The *San Bernardino Mountains Carbonate Plants Draft Recovery Plan*, prepared by the USFWS in 1997, addressed Parish's daisy and four other federally listed species: Cushenbury buckwheat, Cushenbury milk-vetch, San Bernardino Mountains bladderpod (*Physaria kingii* ssp. *bernardina*), and Cushenbury oxytheca (USFWS 1997). The Recovery Plan for these species included the following recovery criteria:

1. Sufficient habitat protected in a reserve system for persistence of existing populations in their ecological context, including the largest populations and best and manageable habitat

2. Identification of potential buffer zones, although not necessarily secured, with an estimate of 4,600 acres needed for habitat connectivity, buffers, and a natural community context
3. Population monitoring and habitat management to provide for early detection of population instability in the reserve system
4. Expansion of existing populations or reintroductions to reduce the chance of extinction due to randomly occurring events.

Based on these recovery criteria, the Recovery Plan identified the following actions:

1. Protect significant extant populations in a reserve system on federally owned land, which would include buffer zones, and maintain selection habitat connections
2. Restore habitat and conduct reintroductions and/or population enhancements where appropriate and feasible
3. Identify and implement appropriate management measures
4. Monitor populations
5. Conduct limited surveys and taxonomic assessments to find new populations.

The Recovery Plan identified the USFS, BLM, California Department of Fish and Game, and USFWS as the agencies primarily involved in the recovery effort (USFWS 1997).

In 2003, the *Carbonate Habitat Management Strategy* (CHMS) was developed by the USFS and BLM in collaboration with a Working Group consisting of mining interests, private landowners, and conservation groups to address impacts to the five federally listed plants associated with carbonate habitats (Olsen 2003). The CHMS, which covers about 160,000 acres (called the Carbonate Habitat Management Area or CHMA), has three main objectives:

1. Economic: regulatory certainty for mining activities, protection of the viability of mining, and streamlining and cost reduction of the permitting process
2. Conservation: maintenance and management of geomorphic and ecological processes of the landscape and placement of habitat

blocks to maintain the carbonate plants, to avoid jeopardy (per Section 7 of the federal Endangered Species Act) and adverse modification or destruction of critical habitat, to contribute to recovery, and to avoid future listings

3. Regulatory: streamlining of permitting, California Environmental Quality Act (CEQA) review, streamlining of County implementation of the California Surface Mining Reclamation Act, and to allow BLM and USFS to comply with certain court-ordered stipulations stemming from lawsuits (i.e., *Center for Biological Diversity v. BLM* and *Southwest Center for Biological Diversity v. Sprague*).

The CHMS includes delineation of an Initial Habitat Reserve, designation of Conservation Units within the CHMA whereby loss and conservation of habitat values can be objectively measured, and contribution by federal agencies and mining interests to reserve assembly through various mechanisms (e.g., dedication of existing unclaimed federal land, purchase of private lands or lands with mining claims, land exchanges, or conservation banking) (Olsen 2003).

Upon successful completion, the CHMS would meet or exceed recovery criteria 1 and 2 listed previously (USFWS 2009).

Implementation of the CHMS has been incorporated by the USFS into the Land Management Plans for the Angeles and San Bernardino National Forests (USFS 2005) and by the BLM into the West Mojave Plan (BLM 2005).

Within the Plan Area, a large percentage of the known populations occur on BLM-administered lands that are covered under the West Mojave Plan (BLM 2005). However, it is estimated by the USFWS that 73% of these lands are under claim to mining companies and development of these sites will make conservation difficult (Sanders 2006). One population around Three Sisters Peak West is under non-profit control, which presumably will have conservation benefits for the species.

Data Characterization

The general distribution of Parish's daisy is fairly well known, based on its close association with carbonate substrates and increased survey efforts since its federal listing as endangered in 1994 (67 FR

78570–78610). However, its population status in terms of population trends is not well understood due to subjective mapping of occurrences between the different survey efforts and a lack of systematic studies carried out over time (USFWS 2009).

Management and Monitoring Considerations

To achieve species recovery, the USFWS (2009) has identified several management and monitoring strategies that need to be implemented for Parish's daisy. These strategies include:

1. Working with the San Bernardino National Forest to conduct systematic monitoring of Parish's daisy throughout known and potentially occupied sites
2. Within occupied Parish's daisy habitat continue monitoring programs for the effectiveness of measures to protect the species from recreation activities
3. Avoid new developments in or near Parish's daisy habitat.

Research by Mistretta and White (2001) indicates that restoration of Parish's daisy population can be successful. A total of 66% of plants transplanted to a disturbed but irrigated site in 1991–1992 survived a 6-year monitoring period. In addition, successful recruitment of progeny was reported at the restoration site. Sanders (2006) suggests that Parish's daisy may be better able to recover after disturbance than some carbonate endemics.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Parish's daisy, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 187,517 acres of modeled suitable habitat for Parish's daisy in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat for Parish's daisy in the Plan Area.

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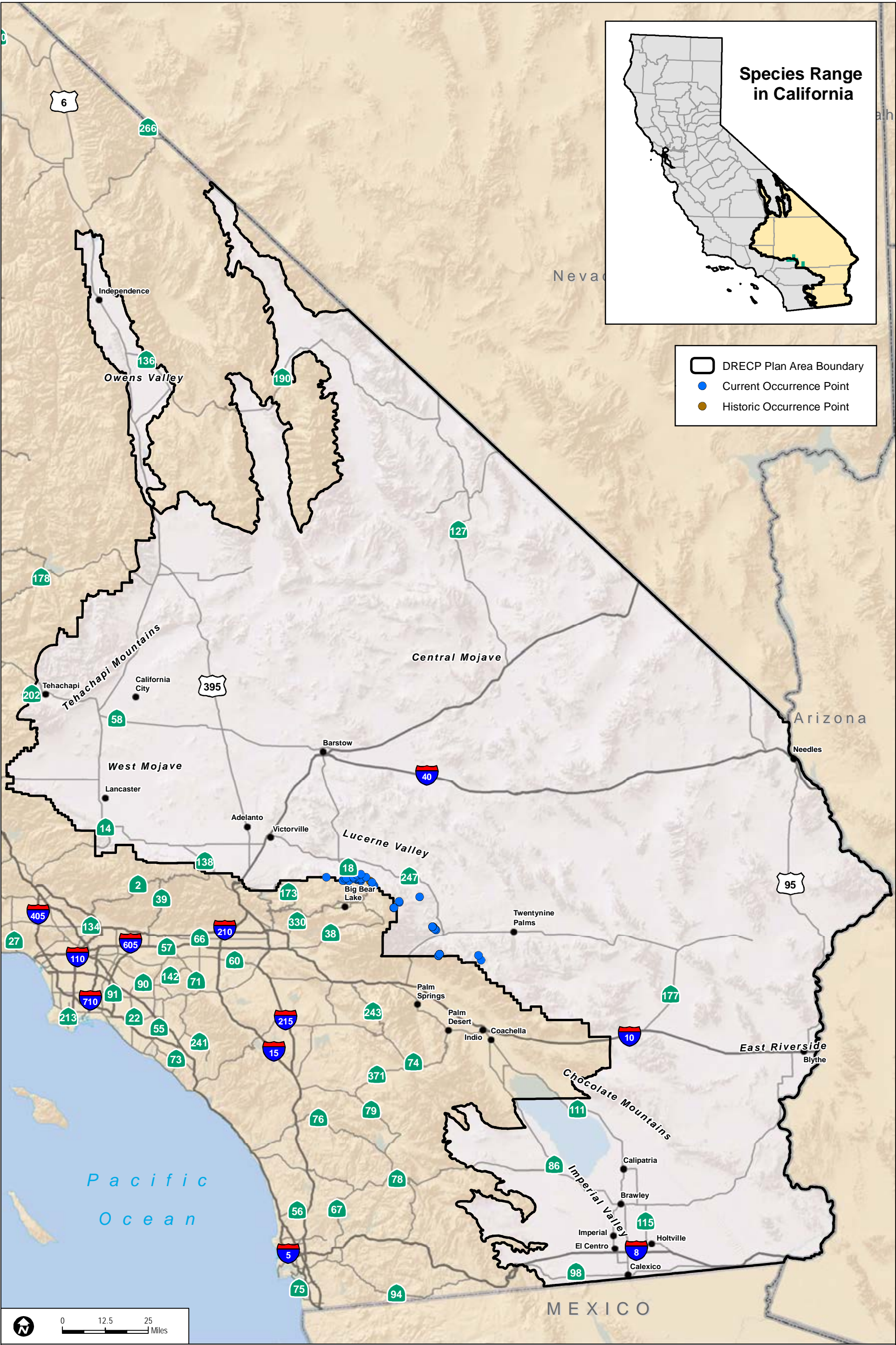
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PLANTS

Parish's Daisy (*Erigeron parishii*)

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P09
Parish's Daisy Occurrences in the Plan Area

Triple-Ribbed Milk-Vetch (*Astragalus tricarinatus*)

Legal Status

State: S1.2¹

California Rare Plant

Rank: 1B.2²

Federal: Endangered; U.S.
Forest Service Sensitive

Critical Habitat: N/A

Recovery Planning: N/A

Notes: The federal 5-year review of the species recommended no change needed for the endangered status of the species (USFWS 2009).



Photo courtesy of John Green.

Taxonomy

Triple-ribbed milk-vetch (*Astragalus tricarinatus*) was first described by Asa Gray in 1876, based on a collection from Whitewater Canyon (63 FR 53596–53615). Although it was transferred to another genus—*Hamosa*—in 1927, this species is currently accepted as *Astragalus tricarinatus* and there is no available information to suggest that the taxonomy of triple-ribbed milk-vetch is uncertain or in question (Jepson Flora Project 2011).

Triple-ribbed milk-vetch is a short-lived, perennial herb with stems approximately 5 to 25 centimeters (2 to 10 inches) in length. A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011).

Distribution

General

The general range of triple-ribbed milk-vetch includes the eastern San Bernardino Mountains/Whitewater Canyon area, Morongo Canyon, and the western part of the Little San Bernardino Mountains, with

¹ **S1:** Critically imperiled; **X.2:** Threatened.

² **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly endangered in California.

disjunct occurrences in the Orocopia (Barneby 1959) and Santa Rosa mountain ranges (Figure SP-P10), although the Orocopia occurrence is unvouchered (USFWS 2009). Throughout the species' range, there are 21 occurrences, of which, 19 are considered extant (CDFW 2013a). Within the Plan Area, triple-ribbed milk-vetch occurs in the Morongo Canyon area and in the Little San Bernardino Mountains at Coyote Hole Spring, Long Canyon, and possibly at Keys Ranch.

Distribution and Occurrences within the Plan Area

Historical

Historically (prior to 1990), triple-ribbed milk-vetch was known from Whitewater and Morongo canyons in Riverside and San Bernardino counties and southeast to the Orocopia Mountains in Riverside County (63 FR 53596–53615). The California Natural Diversity Database (CNDDDB) includes no historical occurrences in the Plan Area (CDFW 2013a). A 1926 collection from a small population is also noted from Coyote Hole Spring along the northern edge of the Little San Bernardino Mountains and south of the town of Joshua Tree (USFWS 2009), but no recent information is available for this site, and the occurrence is not in the CNDDDB (CDFW 2013a). The Keys Ranch site in Joshua Tree National Park is also from 1926 but it was not detected in a 1999 survey (USFWS 2009).

Recent

This description of recent occurrences is primarily taken from the 2009 5-year review of triple-ribbed milk-vetch (USFWS 2009) because it includes all of the CNDDDB occurrences in the Plan Area as well as some occurrences that are not in the CNDDDB. As shown in Figure SP-P10, there are 21 recent occurrence locations for triple-ribbed milk-vetch in the Plan Area: Wathier Landing, Catclaw Flat, Mission Creek, Dry Morongo Canyon and Wash, Big Morongo Canyon, Long Canyon, Coyote Hole Spring, Key's Ranch (note that this site is unvouchered), and Orocopia Mountains. The characterization of the species' distribution is complicated by the fact that the occurrences appear to represent different types of populations: source populations, waifs (i.e., isolated plants), and deme populations (i.e., groups of isolated plants) (USFWS 2009). Source populations are larger, permanent populations (i.e., up to several hundred individuals) typically located in the upper watershed

areas. Waifs are scattered individuals in washes downstream of source populations. Deme populations are discrete or isolated groups of waifs that may exhibit intra-population breeding but do not persist. Habitats associated with these population types are discussed in more detail in Habitat Requirements.

There are two recognized source populations in the Plan Area: Wathier Landing and Catclaw Flat. The Wathier Landing population, which is in the Mission Creek drainage just east of Wathier Landing, supported at least 300 aboveground individuals in 2004 (White 2004) and more than 300 adult individuals and many seedlings in 2005 (Amsberry and Meinke 2007). The Catclaw Flat occurrence was first discovered in 2005 about 2.5 miles from the Wathier Landing site and consisted of about 100 individuals, including seedlings (Amsberry and Meinke 2007). Both sites are conserved on private land owned by The Wildlands Conservancy (TWC).

The other occurrences in the Plan Area are considered deme populations that are not self-sustaining (USFWS 2009). Besides the Wathier Landing and Catclaw Flat source populations, the largest documented population was in Big Morongo Canyon; this population numbered less than 50 individuals in 1993, but a survey of the site in 2005 failed to detect the species (CDFW 2013a). One large reproductive individual (but no seedlings) was found in 2005 on a slide of exposed, decomposed granite on the canyon wall in Big Morongo Canyon (Amsberry and Meinke 2007) within the Bureau of Land Management (BLM) Big Morongo Canyon Reserve (CDFW 2013a). Two waif individuals were detected in Long Canyon in Joshua Tree National Park in 2006 (CDFW 2013a).

It should be noted that botanists suspect that more populations of triple-ridged milk-vetch exist on upland slopes in suitable habitat (e.g., rocky, exposed slopes and ridges), but the rugged terrain occupied by this species makes exploration difficult, and small plants tend to blend in with light-colored granitic substrates, making them hard to detect (Amsberry and Meinke 2007).

Natural History

Habitat Requirements

Triple-ribbed milk-vetch is characterized as generally occurring in Joshua tree woodland and Sonoran desert scrub (see Table 1) (CDFW 2013a; CNPS 2011). Throughout its range, it occurs at elevations of 1,300 to 4,000 feet above mean sea level (amsl) (USFWS 2009). Occurrences within the Plan Area occur at 2,300 to 3,700 feet amsl. However, as discussed in Recent Occurrences, populations are characterized as source populations, deme populations, and waifs. The focus of this description is habitat for source populations because they are considered the most important element for the species for conservation purposes. The deme populations and especially the waif populations that likely occur from seedlings washed downstream and downslope from source population are small and not self-sustaining and, therefore, are not as important for conservation and management. These sites are not the primary habitat for the species (Amsberry and Meinke 2007), and these small ephemeral populations likely do not contribute to long-term viability of the species. However, waifs in the Whitewater Canyon wash area are on an eroded talus of the same soil type that occurs in primary habitat for the source populations (Barrows, pers. comm., 2012).

Table1. Habitat Associations for Triple-Ribbed Milk-Vetch

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mojave mixed woody scrub, Sonoran desert scrub	Primary habitat for source populations	Granitic substrates Elevation 1,300 to 4,000 feet amsl	White 2004 Amsberry and Meinke 2007 CDFW 2013a; USFWS 2009

The Wathier Landing source population occurs on an outcrop of metamorphic rock which is weathering into “unproductive-looking” gravelly soil at about 3,700 feet amsl (White 2004). Triple-ribbed milk-vetch was not detected in surrounding granitic slopes or alluvial fans and washes (White 2004). The substrate where the plants were actually detected was largely bare of other species, but associated

plants included giant needlegrass (*Achnatherum coronatum*), California buckwheat (*Eriogonum fasciculatum*), desert ceanothus (*Ceanothus greggii*), tree poppy (*Dendromecon rigida*), bigberry manzanita (*Arctostaphylos glauca*), bitter snakewood (*Condalia globosa*), hairy yerba santa (*Eriodictyon trichocalyx*), and Mojave yucca (*Yucca schidigera*) (Amsberry and Meinke 2007; White 2004). The Catclaw Flat population was located on decomposed granite substrate on an exposed ridge at about 3,400 feet amsl in association with the same plant species as the Wathier Landing site (Amsberry and Meinke 2007).

The unique soil association is a critical component of the species distribution, although the mechanism for that association is unclear. Little else grows on these soils, but whether it is the lack of competition, a unique chemical composition, or the appropriate level of erosion-disturbance that has fostered the plant soil association has yet to be understood. Where that soil occurs, or where similar soil outcrops occur, triple-ribbed milkvetch is often found. In Mission Creek, on these soil types, but in relatively flat terrain, this milkvetch has been observed primarily after a large disturbance (wildfire with firefighting related soil disturbance) (Barrows, pers. comm. 2012).

Triple ribbed milk-vetch generally occurs in dry washes, at the bases of canyon slopes, and on steep scree slopes (USFWS 2009). Generally, primary habitat for source populations in the Plan Area consists of rocky slopes and ridges that are mostly barren. Notably the two source populations are at the two highest elevations of all of the occurrences in the Plan Area, supporting the notion that the large source populations occur in upslope areas in the upper watersheds and the smaller deme populations and waifs occur at lower elevations in downstream washes and downslope (White 2004; USFWS 2009).

Reproduction

Triple-ribbed milk-vetch is a short-lived, perennial member of the pea family (USFWS 2009). Some species-specific life history information is available for this species and comes from a single study of the species conducted in 2005 and 2006 by Amsberry and Meinke (2007) at the two source populations in Wathier Landing and Catclaw Flat.

The blooming season for triple-ribbed milk-vetch is February through May (CNPS 2011). Amsberry and Meinke (2007) found that 62% of sample individuals at Wathier Landing were in flower in March 2005, and 38% were beginning to produce fruit. At Catclaw Flat, all sampled plants were in fruit in May 2005. Sampled plants at Catclaw Flat reproduced an estimated mean of 2,759 seeds per plant, which is higher than reported rates for other members of this genus. Hundreds of seedlings were observed at both sites in 2005, which was a high rainfall year (a “good” rainfall year), and seedlings were also observed in 2006, which was a dry year. White (2004) also observed seedlings at the Wathier site in 2004, suggesting that reproduction and seedling germination may occur in most years at these source populations (Amsberry and Meinke 2007).

In a pilot greenhouse study of germination requirements of triple-ribbed milk-vetch, Amsberry and Meinke (2007) found that 80% of “viable-appearing” seeds germinated within 72 hours after scarification and wetting; scarification probably occurs naturally through exposure and/or the action of tumbling gravel during flooding. Amsberry and Meinke (2007) also found that growth was more robust in pots inoculated with soil from vigorous, cultivated plants of the obligately mycorrhizal species *Astragalus applegatei* that were previously inoculated with native soil containing mycorrhizae and Rhizobium.

Despite the apparent high productivity of this species, the 5-year review for the species states that “the abundance of this species fluctuates from year to year and may not be present above ground in drought years” (USFWS 2009, p. 1). Long-term studies of this species have not been conducted to determine its response to wet and dry cycles.

Amsberry and Meinke (2007) noted that all mature reproductive individuals appeared to be perennial and many had obvious woody bases. The longevity of individuals is suspected to be 3 to 5 years, but long-term studies are needed (Amsberry and Meinke 2007).

Pollinators of triple-ribbed milk-vetch are unknown. Amsberry and Meinke (2007) noted that field conditions were too windy to observe pollinators but indicate that the species’ showy flowers are typical of legumes pollinated by native bees and honeybees.

Dispersal mechanisms are unknown, but observations of many seedlings around mature reproductive plants suggest that dispersal occurs over short distances within the source populations (Amsberry and Meinke 2007; White 2004). The deme populations and waifs probably stem from seeds washed downstream or downslope from the source populations (USFWS 2009; White 2004).

Ecological Relationships

Little is known about the ecological relationships of triple-ribbed milk-vetch. The 5-year review for the species indicates that the individuals may not appear aboveground during drought years (USFWS 2009), but Amsberry and Meinke (2007) suggest that reproduction and seedling germination may occur in most years at the source populations. Long-term studies are needed to understand the species' response to wet and dry cycles.

The pilot greenhouse study by Amsberry and Meinke (2007) found a positive growth response in soils from the obligately mycorrhizal congener *Astragalus applegatei*, raising the potential importance of relationships with fungal or bacterial associates.

Pollination and dispersal studies have not been conducted, although the species' showy flowers may attract native bees and honeybees, and seedlings are readily observed around source populations (Amsberry and Meinke 2007; White 2004).

Associated plants at the two source populations in the Plan Area—Wathier Landing and Catclaw Flat—are similar, but this similarity is not unexpected because of the close proximity of the two sites. The plant communities at most other occurrences have not been described, but the vegetation community at the East Deception Creek site, which is a deme population of about 50 individuals on a scree slope, includes creosote bush (*Larrea tridentata*), Schott's indigobush (*Psoralea schottii*), rush milkweed (*Asclepias subulata*), burrobrush (*Ambrosia salsola* var. *pentalepis*), and deerweed (*Acmispon glaber*) (Le Doux 2007, cited in USFWS 2009). Given that most occurrences of triple-ribbed milk-vetch are in barren areas, local plant associations do not appear to be an important factor for presence or absence.

Population Status and Trends

Global: G1, Critically Imperiled (NatureServe 2011, Conservation Status last reviewed 2003)

State: S1, Critically Imperiled (CDFW 2013b)

Other than the site-specific counts and population estimates for the approximately 18 extant occurrences for triple-ribbed milk-vetch, there are little data for population status and trends. For the 5-year review of the species, the U.S. Fish and Wildlife Service (USFWS) estimated the known rangewide population to be less than 500 individuals, including source and deme populations and waifs (USFWS 2009). The two observed source populations in the Plan Area—Wathier Landing and Catclaw Flat—were known to support approximately 300 and 500 individuals, respectively, in the mid-2000s (Amsberry and Meinke 2007), but their current status is unknown. The other occurrences in the Plan Area are small, unsustainable deme populations and waifs (see Recent Occurrences). However, the actual population is likely to be substantially larger because not all suitable habitat areas have been surveyed. The observed deme populations and waifs in downstream and downslope areas indicate the likely presence of larger, but as yet unknown, upslope source populations (USFWS 2009).

Threats and Environmental Stressors

The main anthropogenic threats to triple-ribbed milk-vetch that triggered the federal listing of the species in 1998 was bulldozing for maintenance of a gas pipeline and earth-moving activities along a stretch of Big Morongo Canyon to realign segments of a crude oil pipeline that had been exposed during winter storms in 1992–1993 (63 FR 53596–53615). It is considered to be under continuing threat from maintenance of the crude oil pipeline and from off-highway vehicle use in the canyons. Its small population numbers make it vulnerable to stochastic events and anthropogenic events such as pipeline leaks (USFWS 2009). New threats identified since the species' federal listing include wildland fire suppression activities, flooding, and climate change (USFWS 2009). Amsberry and Meinke (2007) also identify exotic weed infestations resulting from increased vehicle and foot traffic as a potential threat to the species.

Rangewide, but outside the Plan Area, other potential threats include residential development of population location in East Deception Canyon and Lower Mission Creek, which may affect downstream habitat and facilitate off-highway vehicle use (USFWS 2009).

Conservation and Management Activities

Conservation and management activities within the Plan Area include preservation of the two known source populations—Wathier Landing and Catclaw Flat—on the Whitewater Preserve, privately owned by TWC. These lands are operated and managed with the same goals as the surrounding BLM San Gorgonio Wilderness Area (USFWS 2009). TWC also leased a nearby 40,032-acre BLM grazing allotment that has since been relinquished, and grazing is no longer permitted (USFWS 2009).

Small populations of triple-ribbed milk-vetch occur in Big Morongo Canyon in the Plan Area within the BLM Big Morongo Canyon Preserve, which is designated an Area of Critical Environmental Concern (ACEC), encompassing about 31,000 acres. Further, the San Gorgonio Additions Wilderness Area comprises approximately 39,215 acres between San Bernardino National Forest and the Morongo Valley; it includes significant portions of the Mission Creek and Whitewater drainages, and preserves significant contiguous occurrences and contiguous habitat (USFWS 2009). It is highly possible that additional source populations within the Plan Area occur in the San Gorgonio Additions Wilderness Area, given the nearby locations of the Wathier Landing and Catclaw Flat source populations (see Figure SP-P10).

The Long Canyon and Keys Ranch occurrences are within Joshua Tree National Park, and as of 2009 a management plan was being prepared for the species (USFWS 2009).

Conservation of the species outside the Plan Area is provided by the Coachella Valley Multiple Species Habitat Conservation Plan (MSHCP), which conserves 2,838 of the 3,007 acres of modeled habitat distributed across Whitewater Canyon (1,295 acres), Mission Creek and Big Morongo Canyon (819 acres), Whitewater floodplain (866 acres), and Santa Rosa and San Jacinto Mountains (1 acre) (CVMSHCP 2007).

Data Characterization

The geographic range of triple-ribbed milk-vetch probably is fairly well known since no new outlier populations have been discovered since 1985 (the Agua Alta site in the Santa Rosa Mountains). The Orocochia Mountains occurrence is unvouchered. However, within the species' geographic range boundaries, its distribution probably is still not well understood. Only two source populations for the species that are in close proximity to each other have been documented—the Wathier Landing and Catclaw Flat occurrences on TWC land. The other documented occurrences are deme populations and waifs that indicate a larger upslope source population that has not been documented but provides seedlings for the downstream and downslope populations (USFWS 2009). Because of the rugged and potentially inaccessible primary habitat for the species (i.e., rocky slopes, canyon walls, and ridges in remote upper watershed areas), much suitable habitat probably has not been adequately surveyed. In addition, if the species' abundance and detectability varies in relation to wet and drought cycles, it may not be detectable on occupied sites in a dry year and follow-up surveys would be required (USFWS 2009). Also, smaller individuals are difficult to detect from a distance because they blend in with the light-colored granitic substrates on which they occur (Amsberry and Meinke 2007). For these reasons, it is likely that the current distribution information significantly underestimates its actual distribution.

Management and Monitoring Considerations

The 5-year review for triple-ribbed milk-vetch (USFWS 2009) recommended several actions related to management and monitoring of the species, including:

- Demographic and survival studies at known sites
- Predictive habitat modeling involving source soils to locate new source populations
- Site-specific fire suppression plans, including avoidance areas, bulldozer lines, and aerial retardant drops, as well as post-fire surveys
- Development of protocols to ensure low impacts during facilities maintenance (e.g., pipelines).

Species Modeled Habitat Distribution

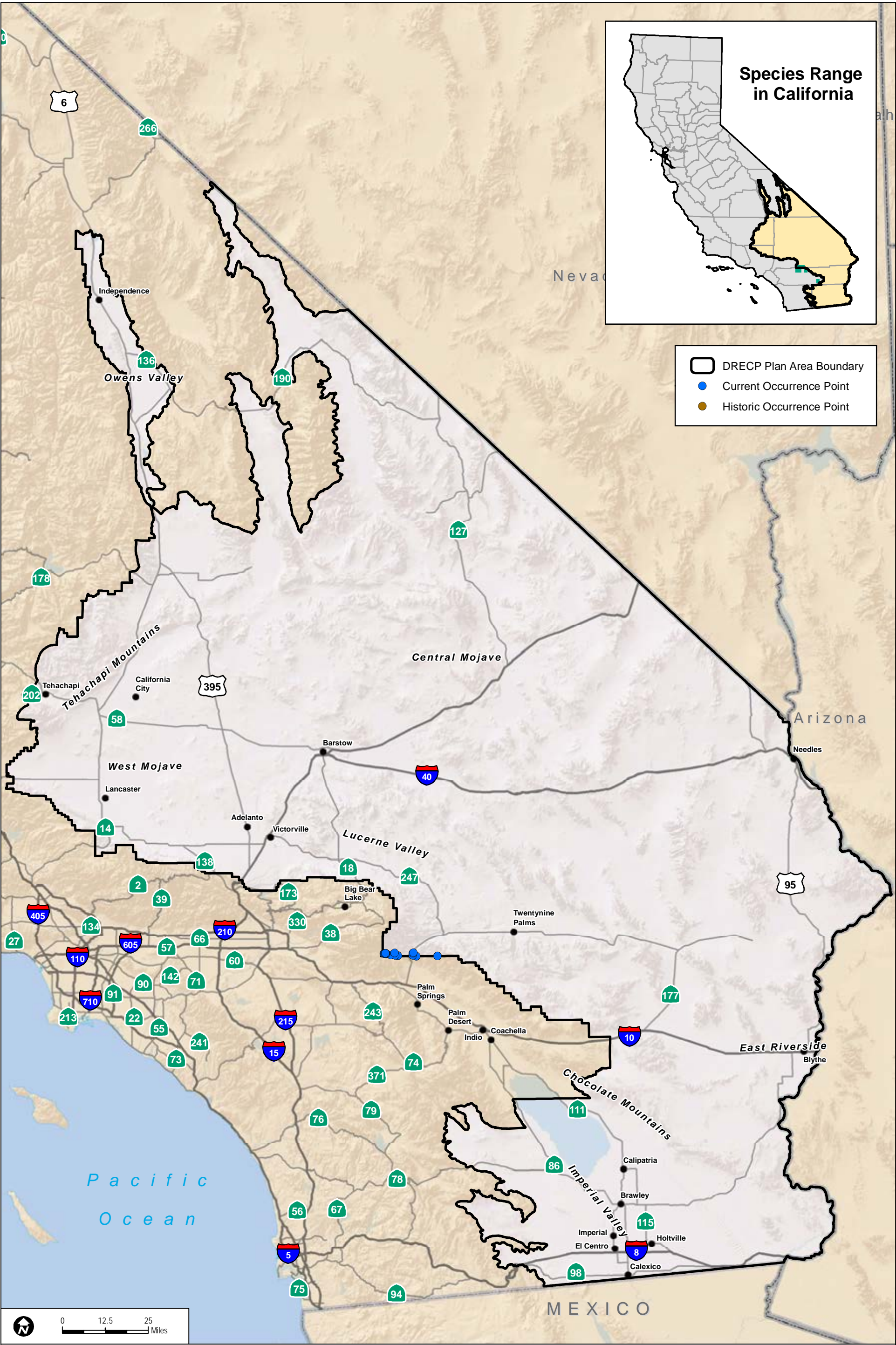
This section provides the results of habitat modeling for triple-ribbed milk-vetch, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 81,251 acres of modeled suitable habitat for triple-ribbed milk-vetch in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P10
Tripple-ribbed Milk-vetch Occurrences in the Plan Area

APPENDIX C

Species Habitat Models

APPENDIX C

Species Habitat Models

This appendix describes the species habitat modeling (also referred to as species distribution modeling) methods for the Desert Renewable Energy Conservation Plan (DRECP) and presents the species habitat model results. The description below provides an overview of the species habitat modeling method that was used to develop the habitat models for each of the proposed Focus Species. Detailed technical information on methods, data, and processing is provided at <http://databasin.org/>.

C.1.0 BACKGROUND

Species habitat modeling (i.e., species distribution modeling) is a necessary component of the planning process for DRECP because of the following factors:

- Need for extrapolating species and habitat distribution across areas lacking adequate data due to lack of comprehensive survey results across the Plan Area;
- Need to obtain information that will supplement existing surveys as part of the planning process;
- Need to transcend the limitations of the “snapshot in time” that survey data represents when using existing field data alone;
- Need for synthesis and analysis of multiple data sources across the entire Plan Area;
- Need to identify and rank biological values between areas; and
- Need to establish baseline conditions to compare alternate conservation strategies.

Given these factors, the DRECP Independent Science Advisors (ISA) “recommend careful use of habitat suitability models or species distribution models” (DRECP ISA 2010). Species habitat modeling can provide an objective, transparent, and repeatable means of assessing species habitat distribution where the species distribution or distribution of suitable habitat for a species is not well known. For these reasons, species habitat modeling results provide additional biological information to be used in the following components of the DRECP: conservation strategy, impact analysis, and monitoring and adaptive management. The approaches to assess the potential effects of climate change on species habitat and distribution for the DRECP are being developed and are not addressed in this document. Additionally, the approaches to address reference states for the purposes of monitoring and adaptive management for the DRECP are being developed and are not addressed in this document.

Generally, two types of models were used for the DRECP: expert-based models and statistically based models. **Expert-based models** identify species-specific habitat distribution based on scientific literature, habitat characteristics, location of documented

APPENDIX C (Continued)

occurrences, and expert opinion related to the physical and biological habitat parameters associated with species occurrence. As the ISA stated, expert-based models are appropriate where species occurrence data are not sufficient (i.e., too few data points to build a model) to conduct more rigorous modeling, where species occurrence data are strongly biased spatially across a plan area, or during the initial, exploratory analyses of environmental factors associated with species occurrence. **Statistically based models** specify suitable habitat and may even predict the likelihood of species occurrence based on correlations between presence/absence data and physical and biological habitat parameters. The ISA indicated that empirical, statistically based models are preferred over expert-based models (such models better control for subjective or biased input). Both expert-based models and statistically based models were developed for proposed Focus Species for the DRECP depending on species-specific considerations, including the availability of data.

The output from statistically based models is a continuous probability value ranging from 0 to 1 corresponding to range from unsuitable conditions for the species to high likelihood of species presence. The output from expert-based models is a binary result indicating suitable habitat or not. In order to use the statistically based models in conjunction with the expert-based models in developing the DRECP, a threshold value was developed for each statistically based model to convert the continuous result into a binary result.

The use of models in the DRECP conservation planning process focused on identifying areas of suitable conditions for a species (i.e., species habitat) within the Plan Area. The statistically based (i.e., Maxent) species distribution models were used in conjunction with the expert-based models to assist in the identification of potential high-priority conservation areas for the DRECP conservation strategy. Models were also used as one measure of quantification of expected conservation and effects for evaluation of conservation strategy alternatives.

C.2.0 SPECIES HABITAT MODEL DEVELOPMENT

Species habitat models have been developed for the 37 proposed Focus Species under the DRECP. The following summarizes the process for developing the DRECP species habitat models.

Early in the DRECP planning process, existing published species distribution models for proposed Focus Species were gathered and evaluated. Additionally, early versions of expert-based and Maxent models were developed for the DRECP. These early model versions were used to support the initial DRECP planning process and were documented in previous versions of the draft Baseline Biology Report (Dudek and ICF 2012) and the Description and Comparative Evaluation of the Draft DRECP Alternatives (DRECP REAT 2012).

APPENDIX C (Continued)

In order to continue to refine and improve the species habitat models, the models documented in Dudek and ICF (2012) went through the following review process:

1. Outside Expert Review (Winter–Spring 2012). This involved the individual review of species profiles and species habitat models by outside scientists and species experts. Comments on profiles have been integrated in the profiles in Appendix B of this document. Comments on species habitat models were used to refine the species habitat models.
2. Independent Science Panel Review (Summer 2012). This involved a panel review of the science used in the DRECP. Comments on species habitat models were used to refine the species habitat models.
3. DRECP Species Modeling Forum (January 2013). Researchers and modelers with expertise in species distribution modeling were gathered with REAT agency biologists to review existing species habitat models and provide species-by-species recommendations on data sources and modeling approaches, as well as address issues common to species modeling in general (including technical issues, such as thresholds, raised in DRECP independent science reviews). For taxa with multiple available models, this forum allowed selection of the one most relevant to the DRECP's purposes and discussion of the differences among the various models for a given taxon. Experts from the Conservation Biology Institute (CBI), University of California Berkeley (UCB), University of California Davis (UCD), University of California Santa Barbara (UCSB), and the United States Geological Survey (USGS) collaborated to develop the recommendations. These scientists also provided recommendations and advice on specific technical issues arising during the DRECP species model development work but subsequent to the forum.

This comprehensive input gathering process provided robust input from species experts, agency specialists, and modelers, and was used to scientifically vet, refine, and improve the DRECP species habitat models for all proposed Focus Species. Statistically based Maxent models were used for a majority of the DRECP Focus Species. Where statistically based models were not recommended due to data limitation or species-specific considerations, expert-based models were developed. Species habitat models used for DRECP were developed by several entities, including CBI, Dudek, UCB, UCD, UCSB, and USGS.

The model results for each species are provided in this appendix. Supporting documentation with detailed information on methods, data, and processing is provided at <http://databasin.org/>.

APPENDIX C (Continued)

C.3.0 REFERENCES CITED

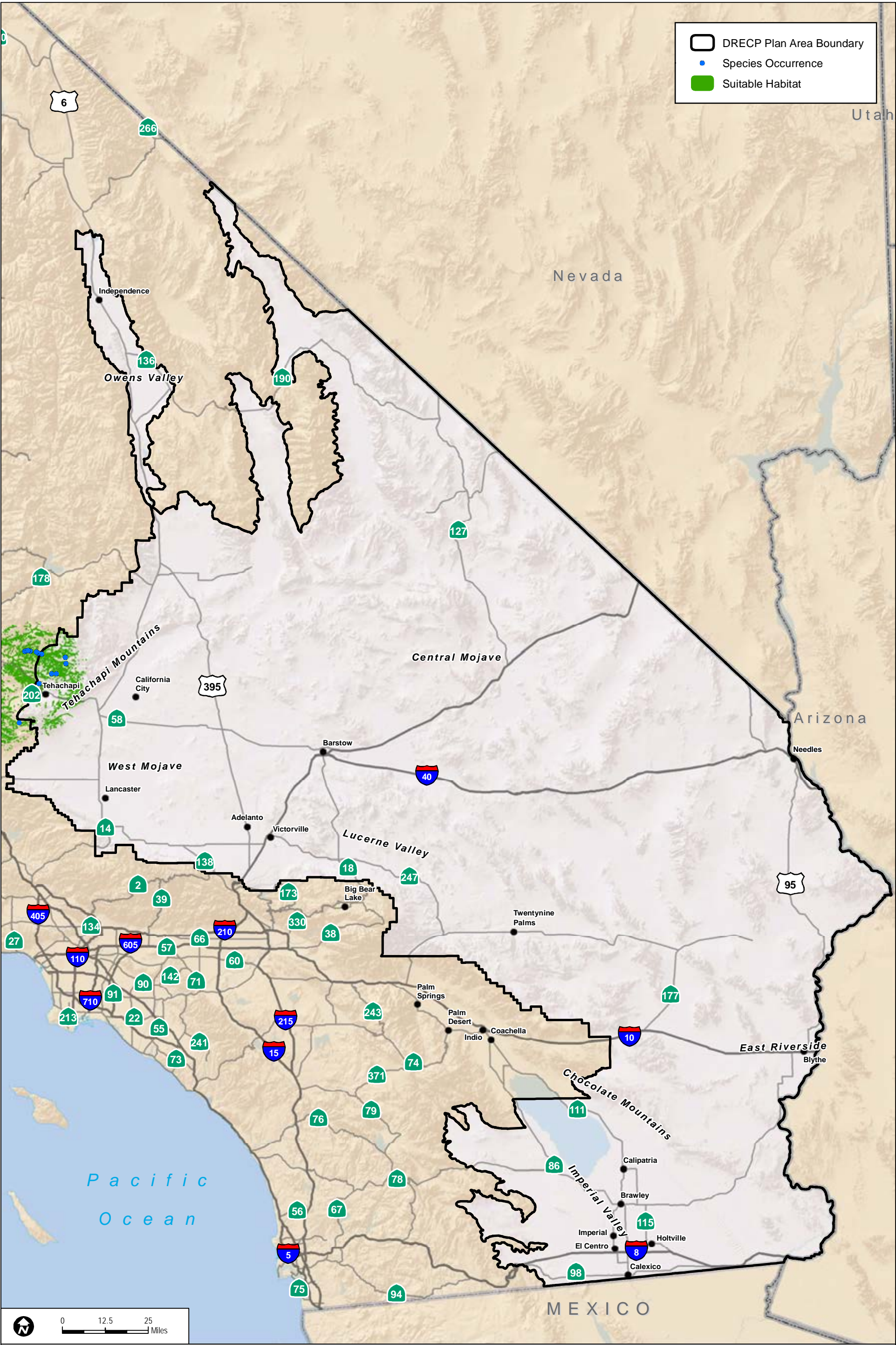
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APPENDIX C (Continued)

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APPENDIX C (Continued)

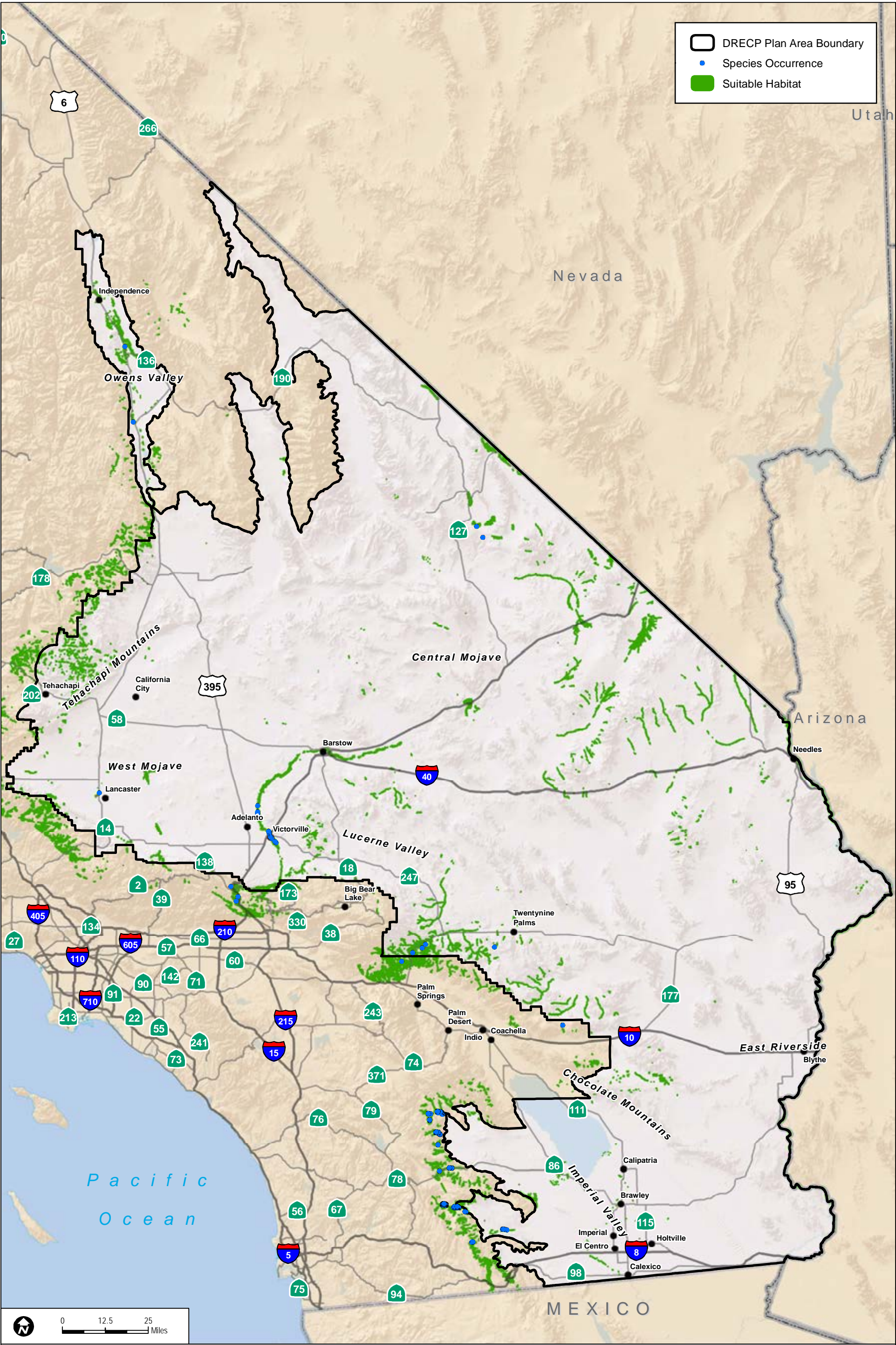
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Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

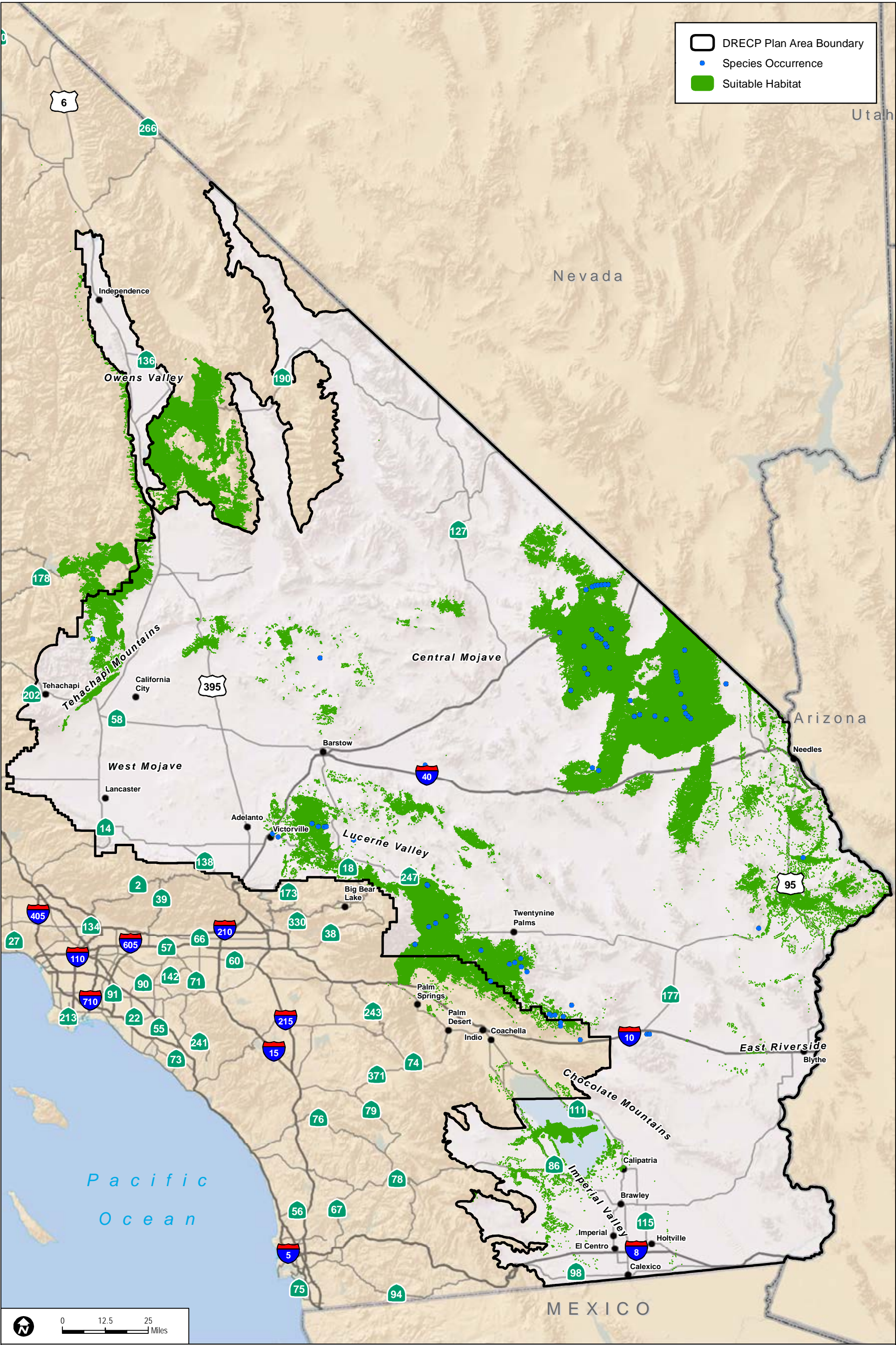
FIGURE SM-A01

Draft Species Habitat Model Results for Tehachapi Slender Salamander



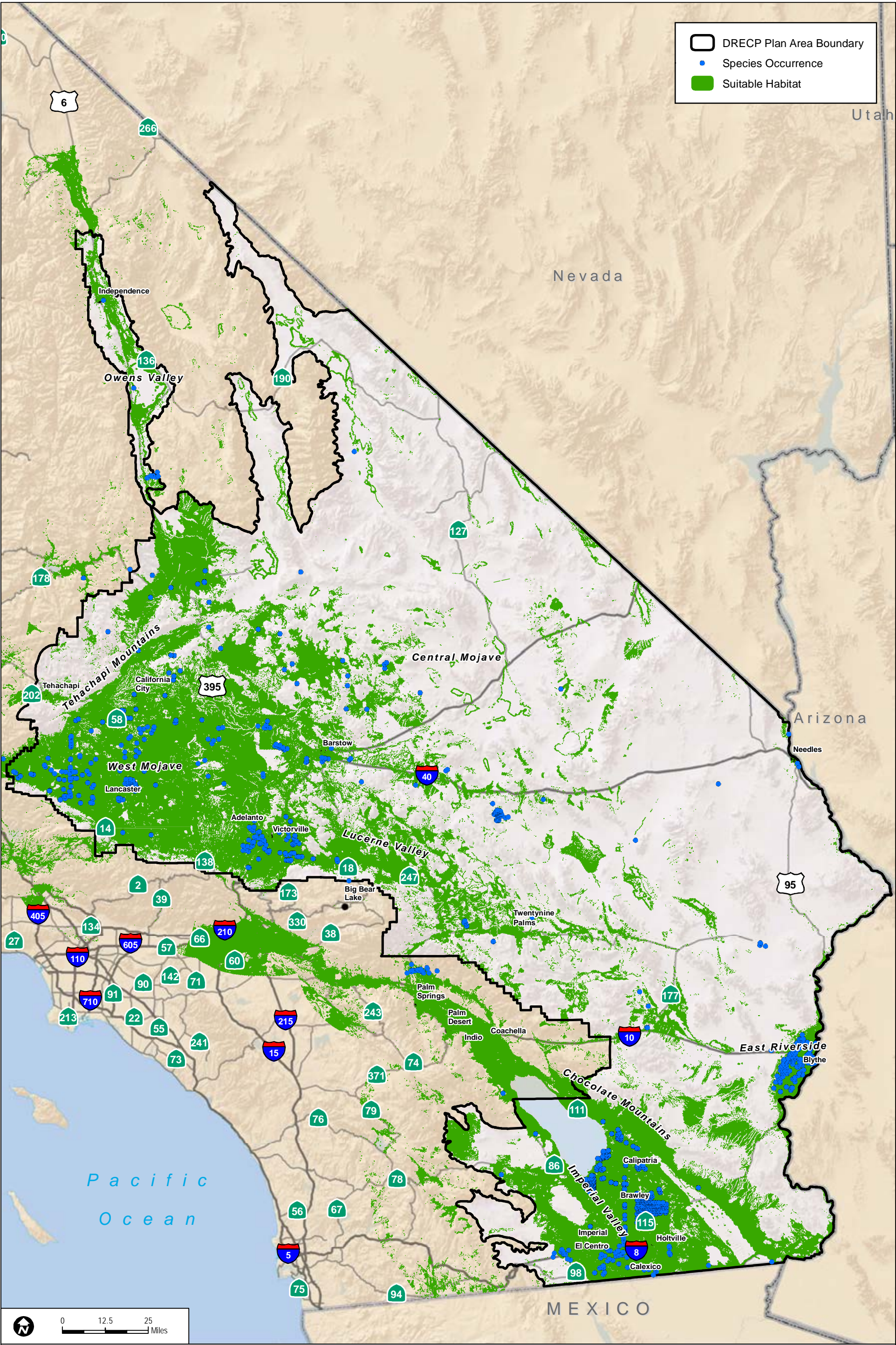
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B01
Draft Species Habitat Model Results for Least Bell's Vireo



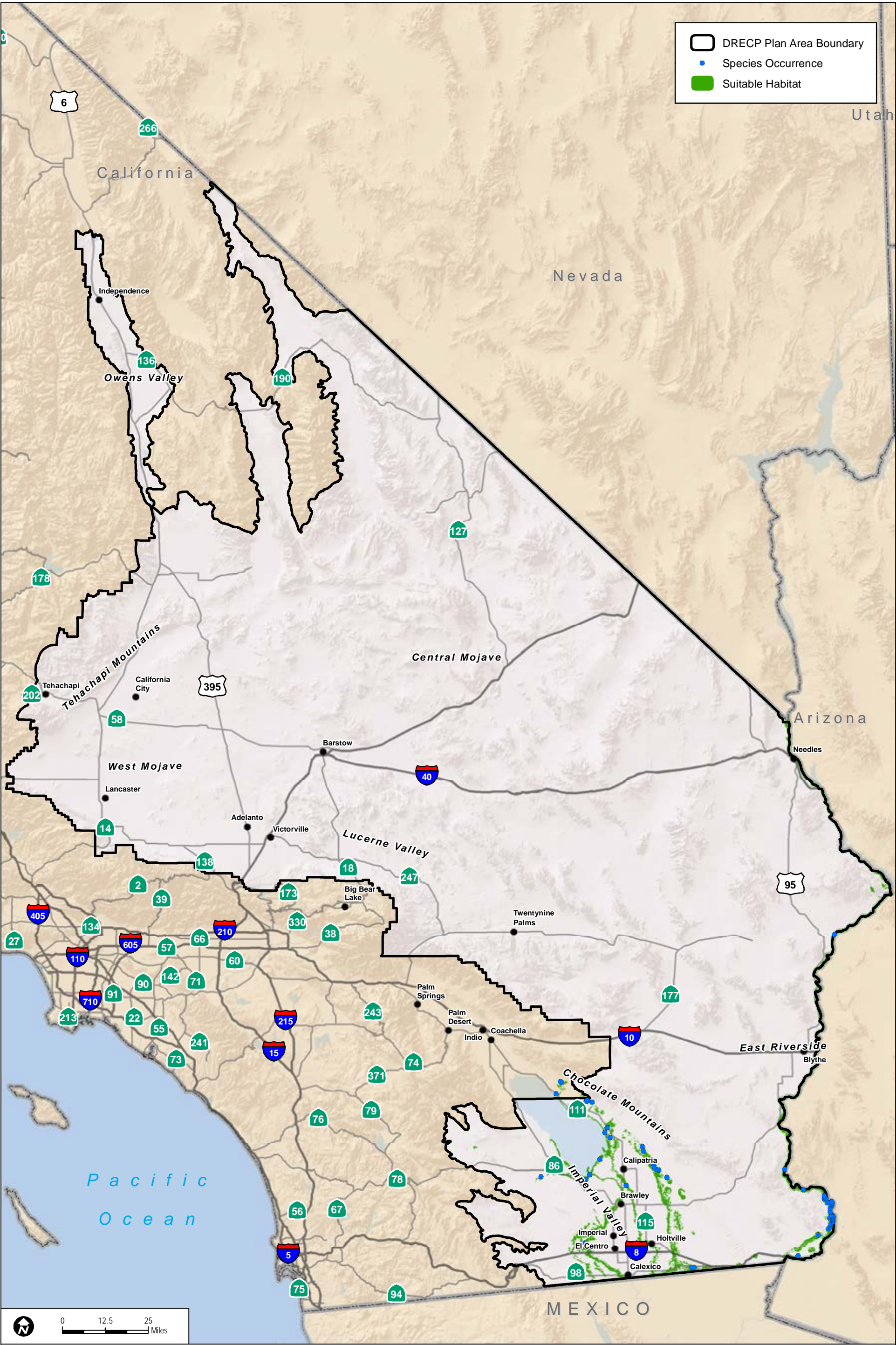
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B02
Draft Species Habitat Model Results for Bendire's Thrasher



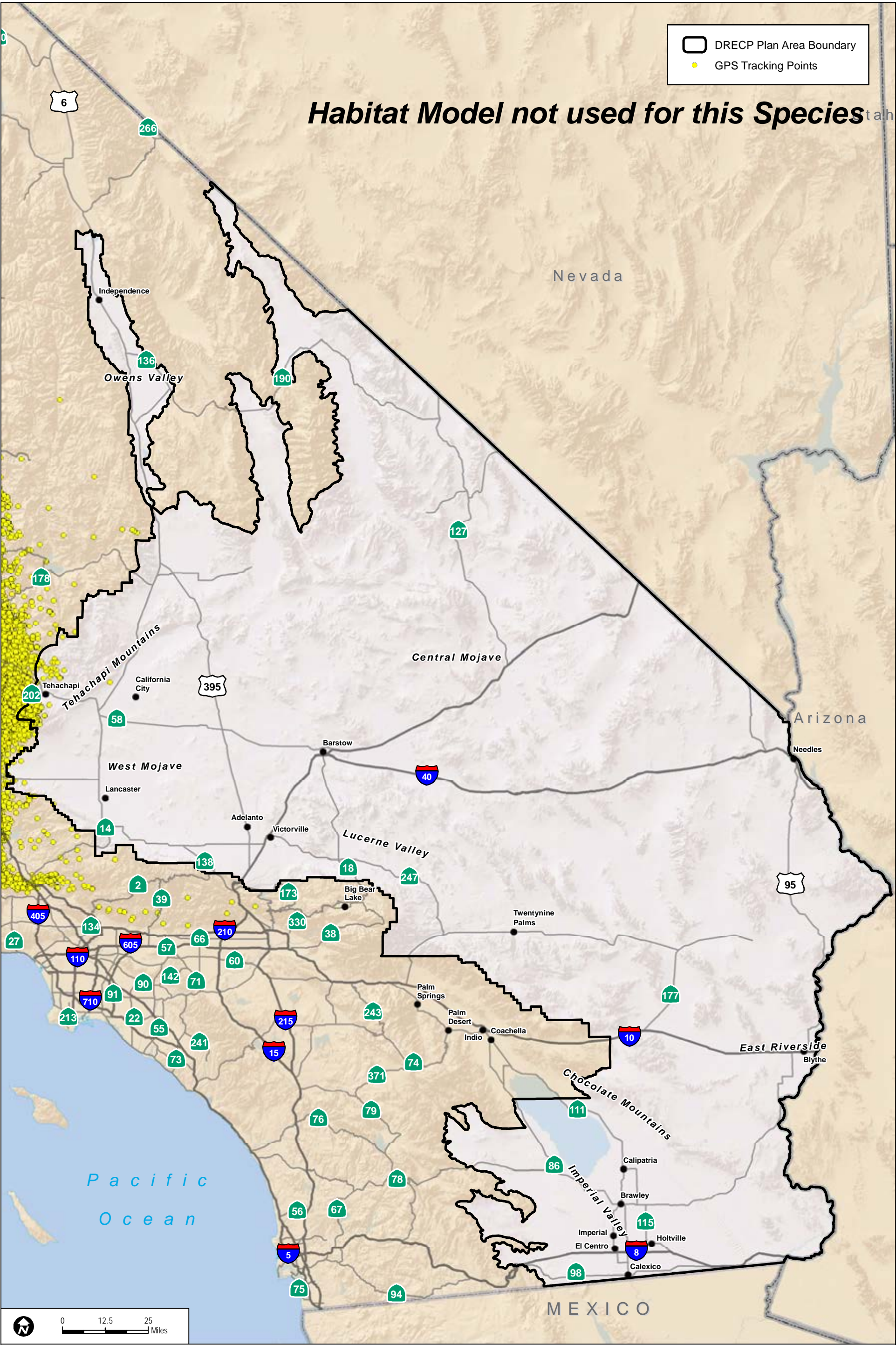
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B03
Draft Species Habitat Model Results for Burrowing Owl



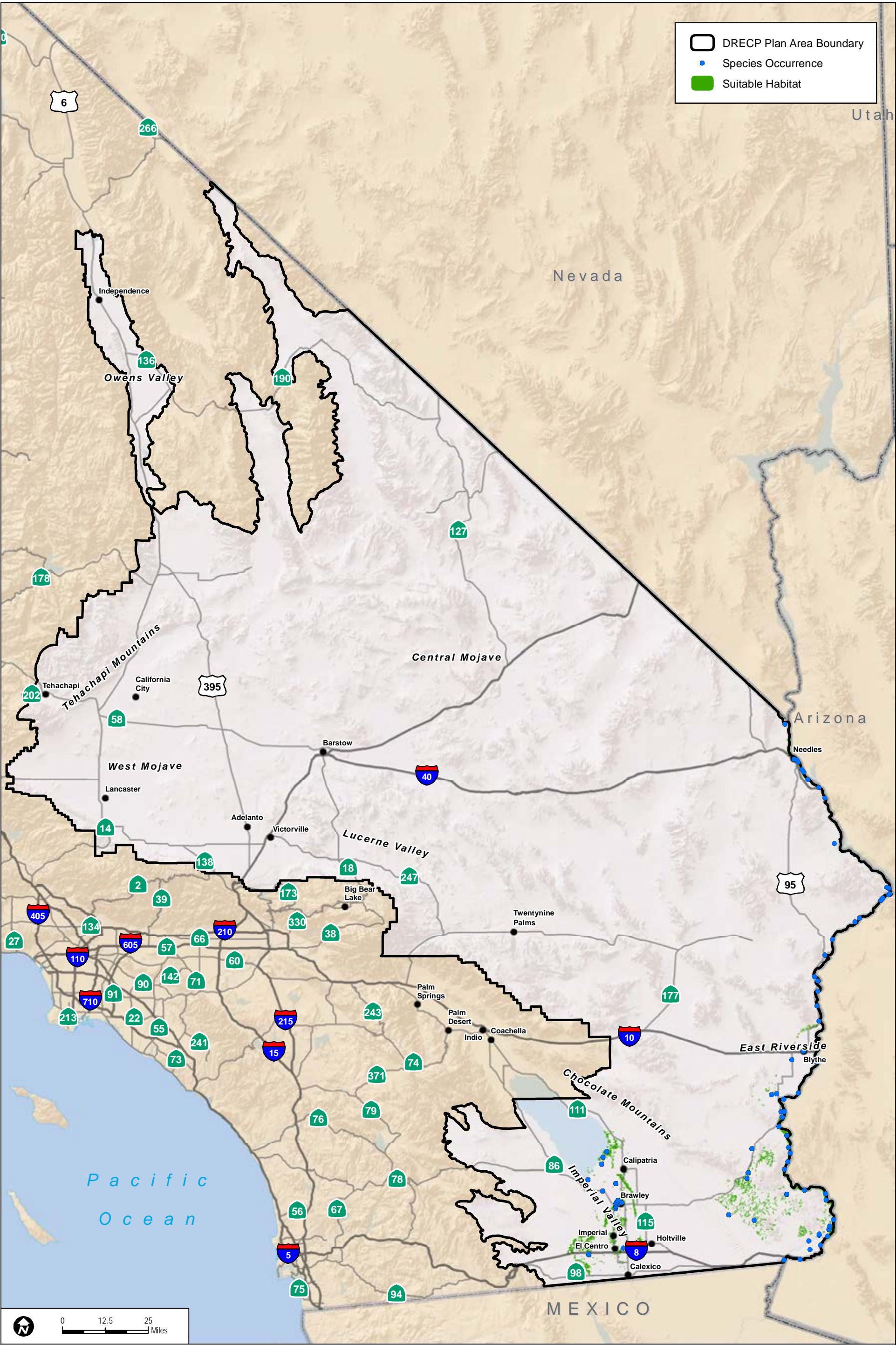
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B04
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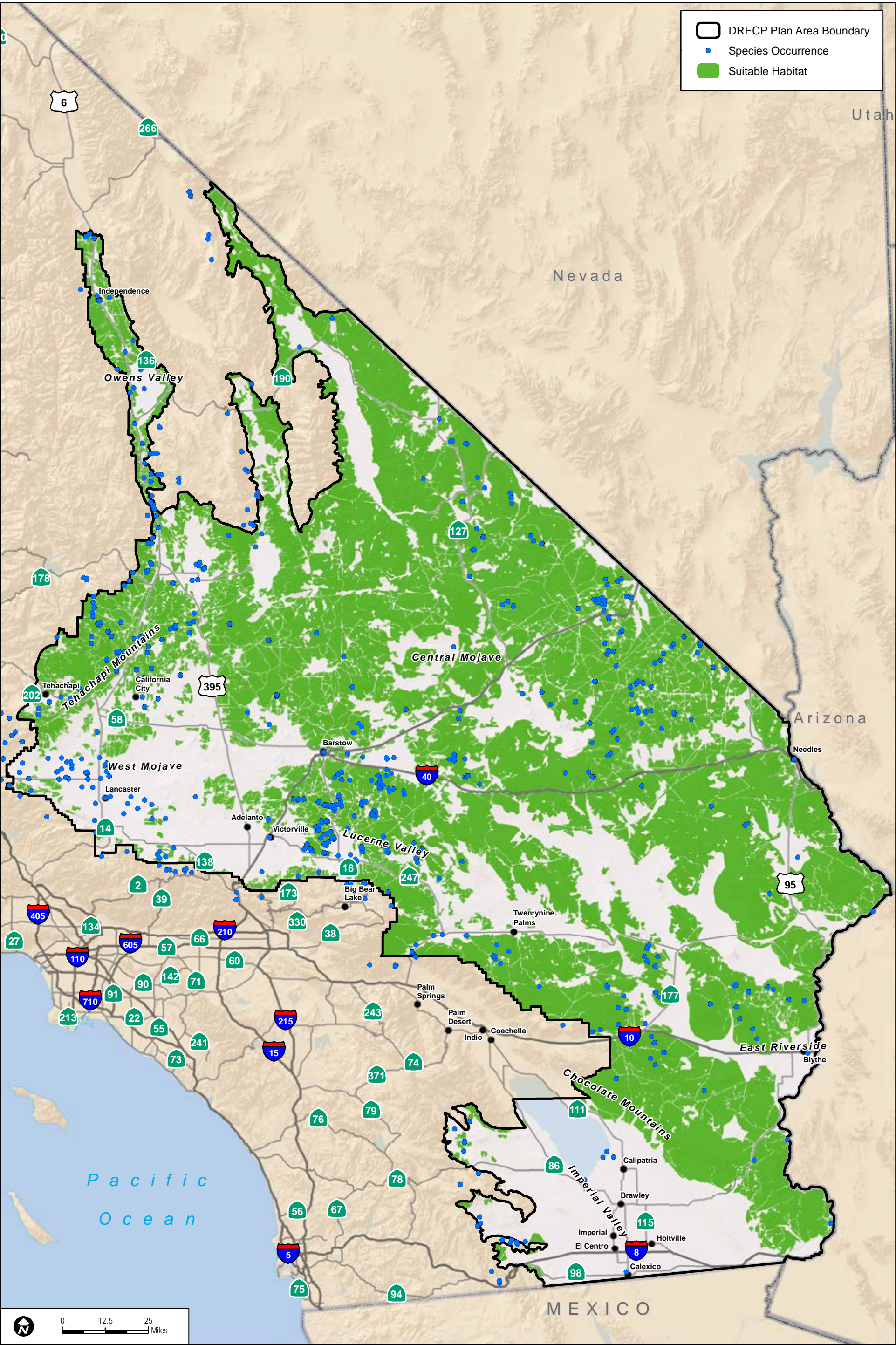
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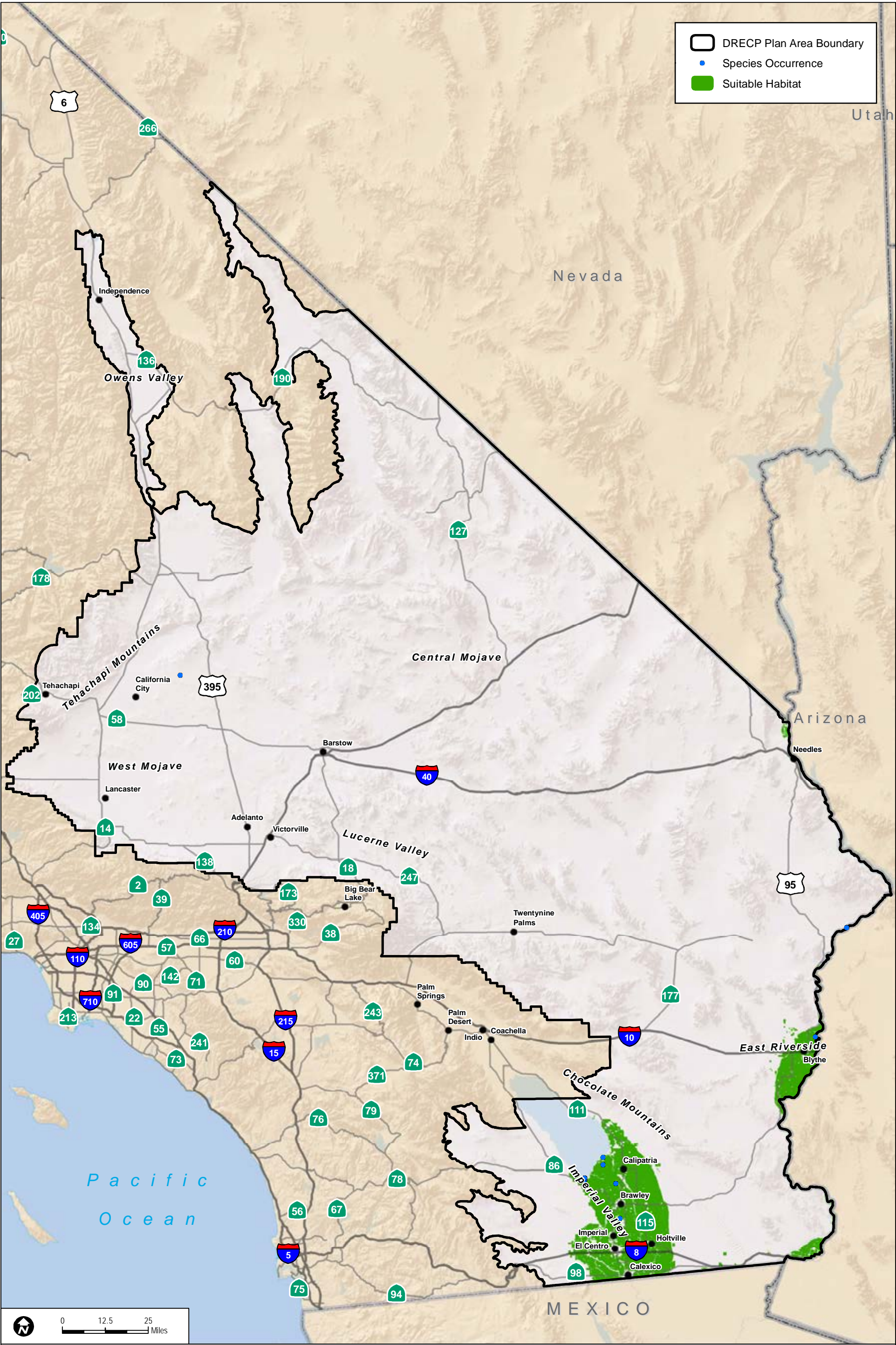
FIGURE SM-B05
Draft Species Habitat Model Results for California Condor



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B06
Draft Species Habitat Model Results for Gila Woodpecker

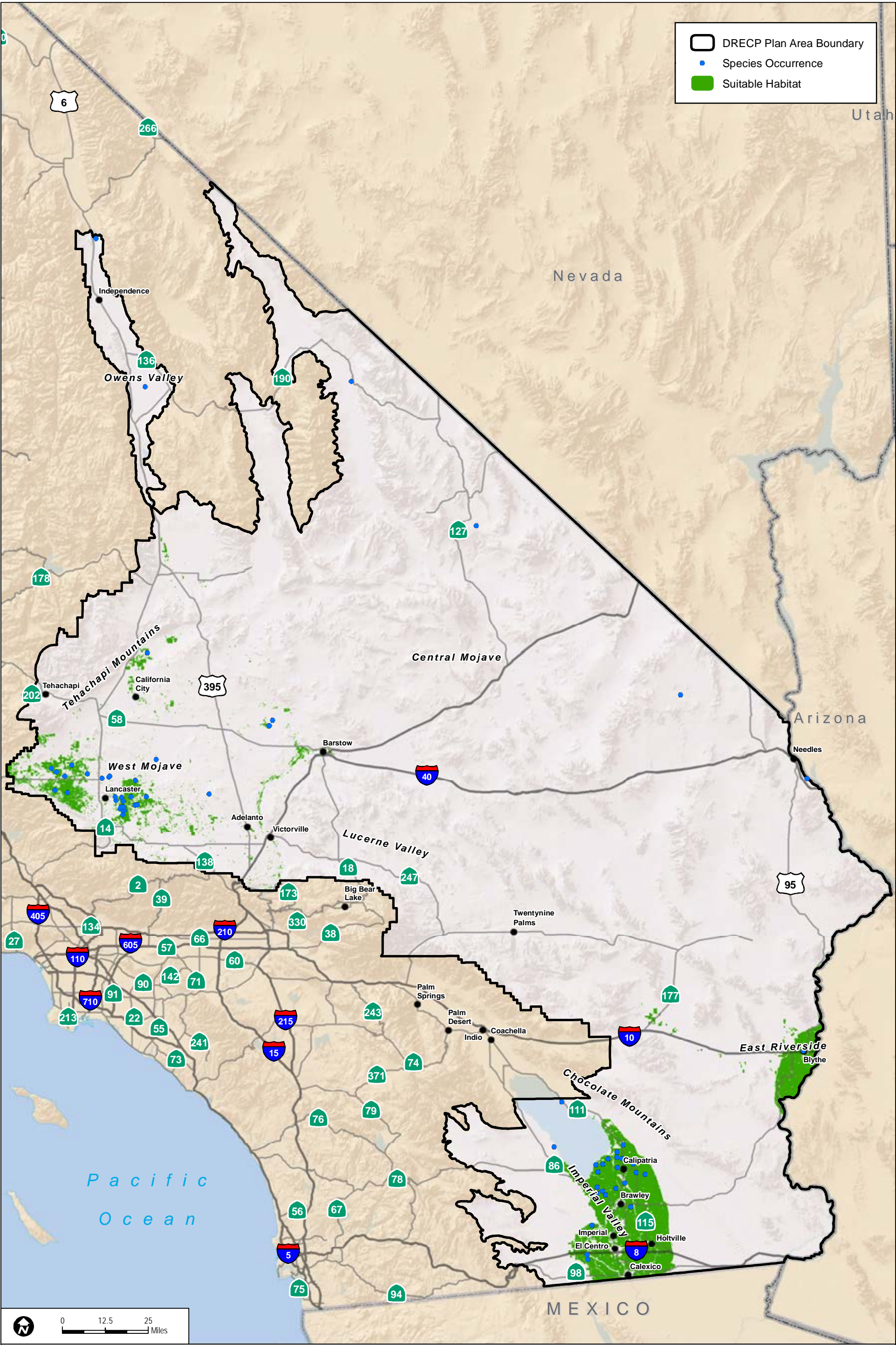




Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

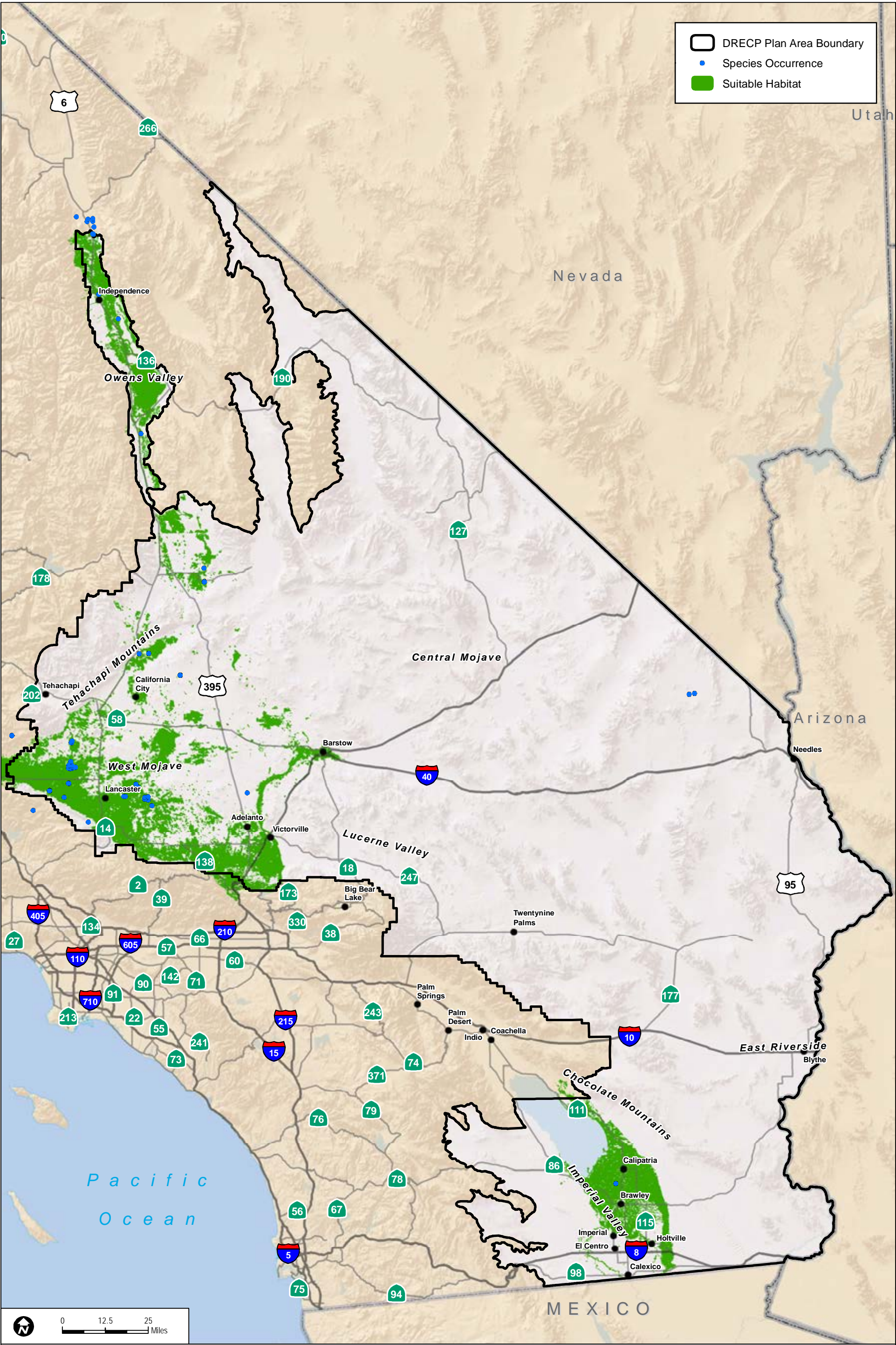
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Draft Species Habitat Model Results for Greater Sandhill Crane



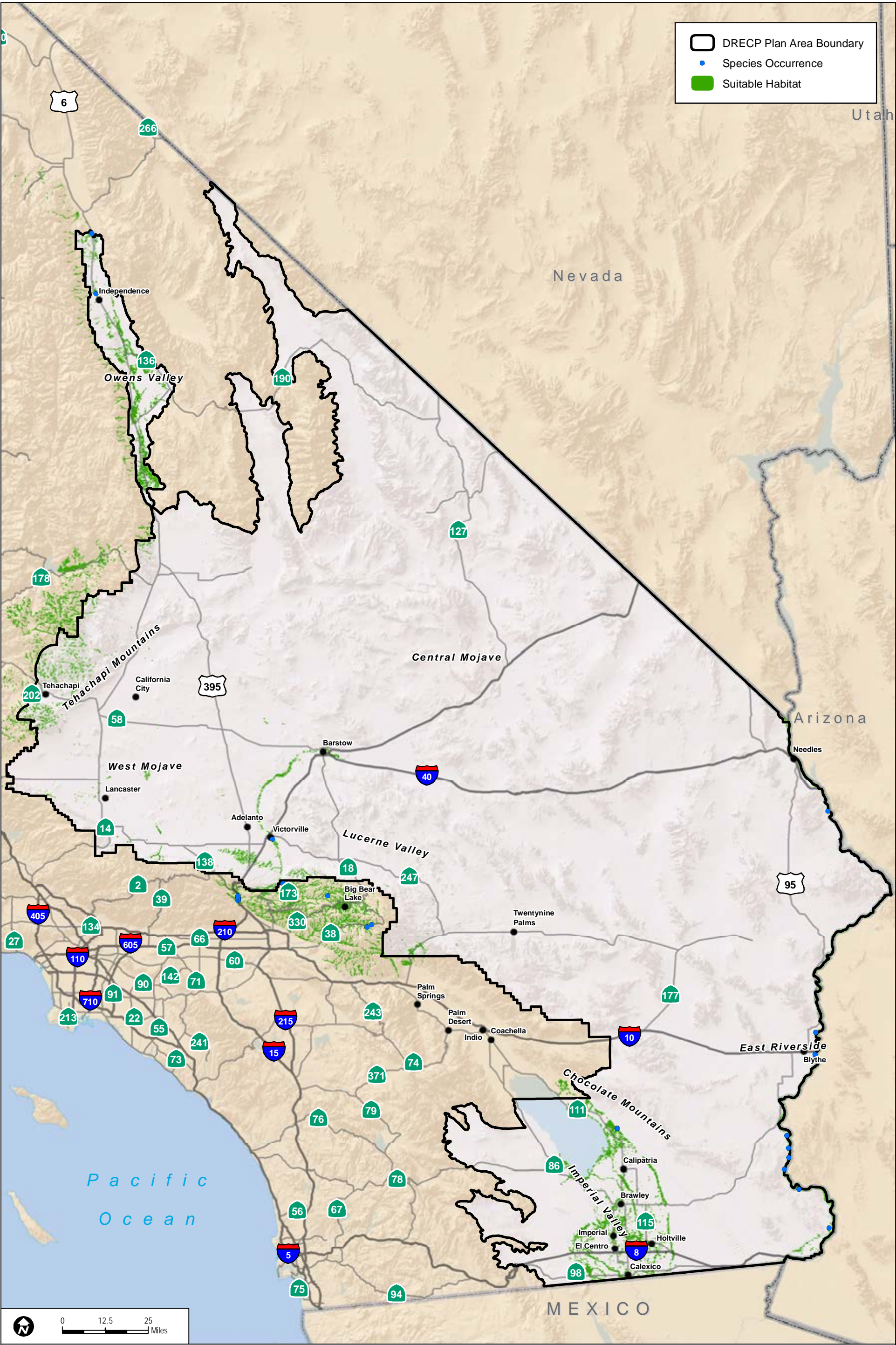
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B09
Draft Species Habitat Model Results for Mountain Plover



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

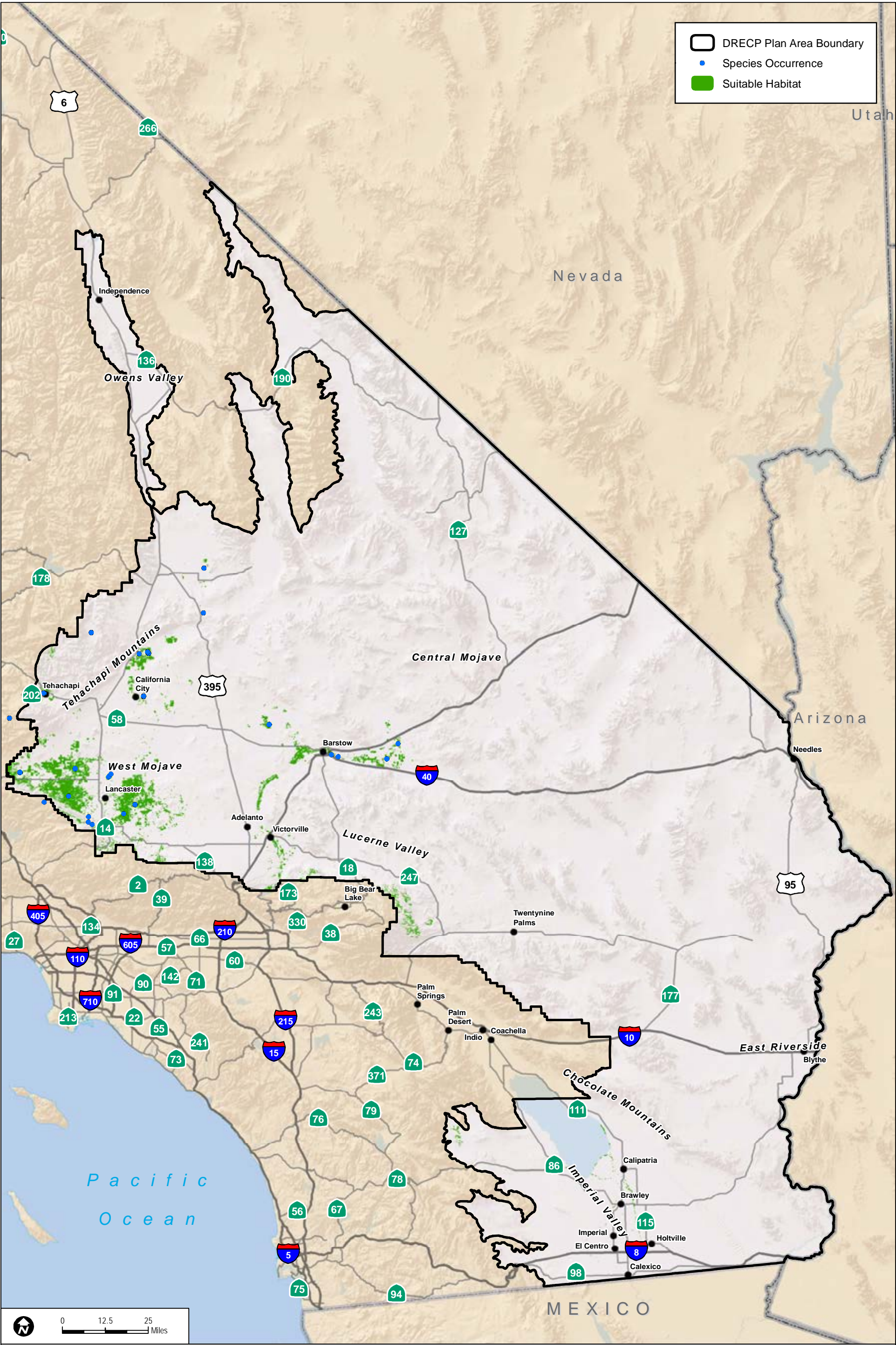
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Draft Species Habitat Model Results for Swainson's Hawk



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

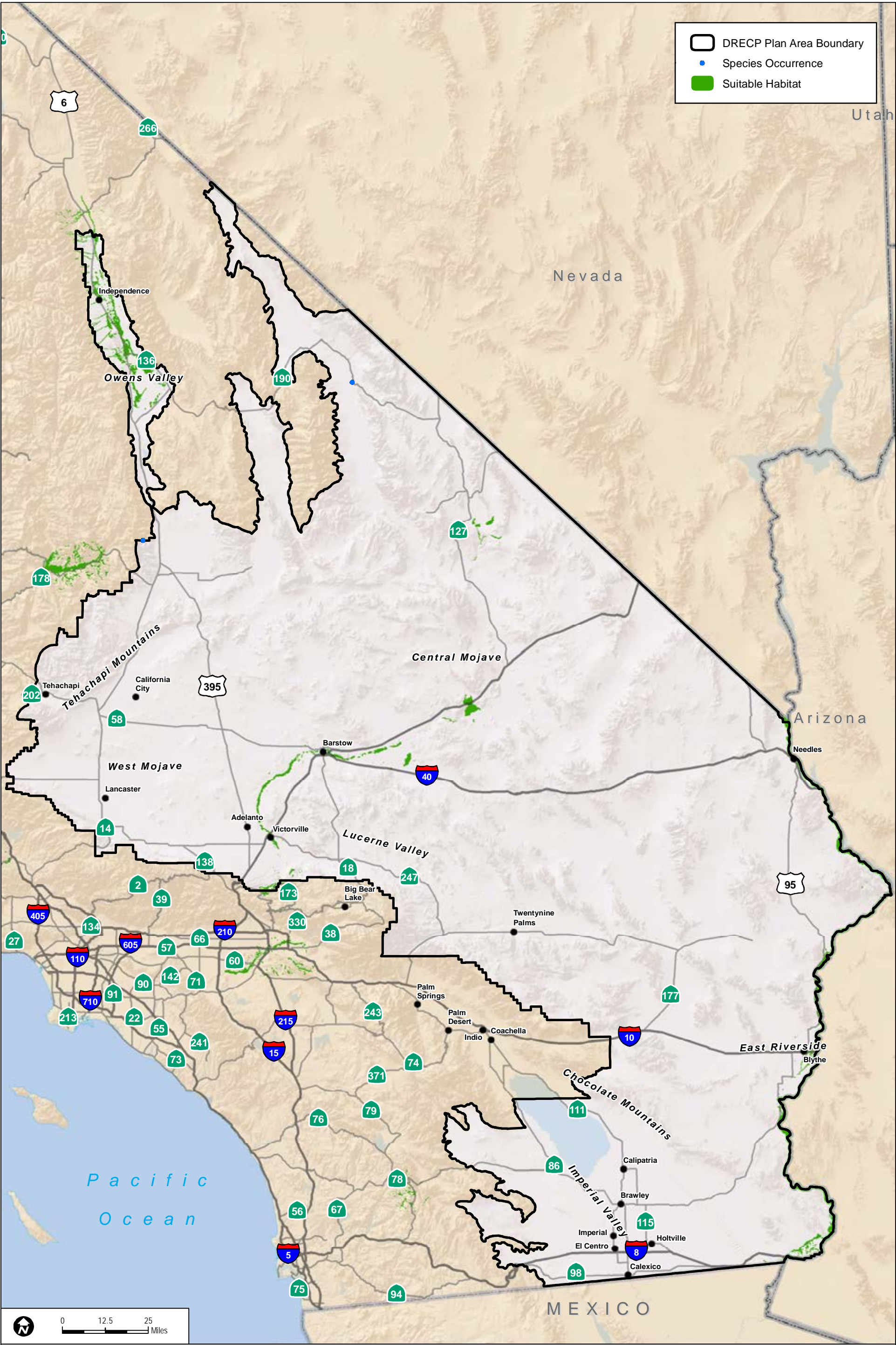
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Draft Species Habitat Model Results for Southwestern Willow Flycatcher



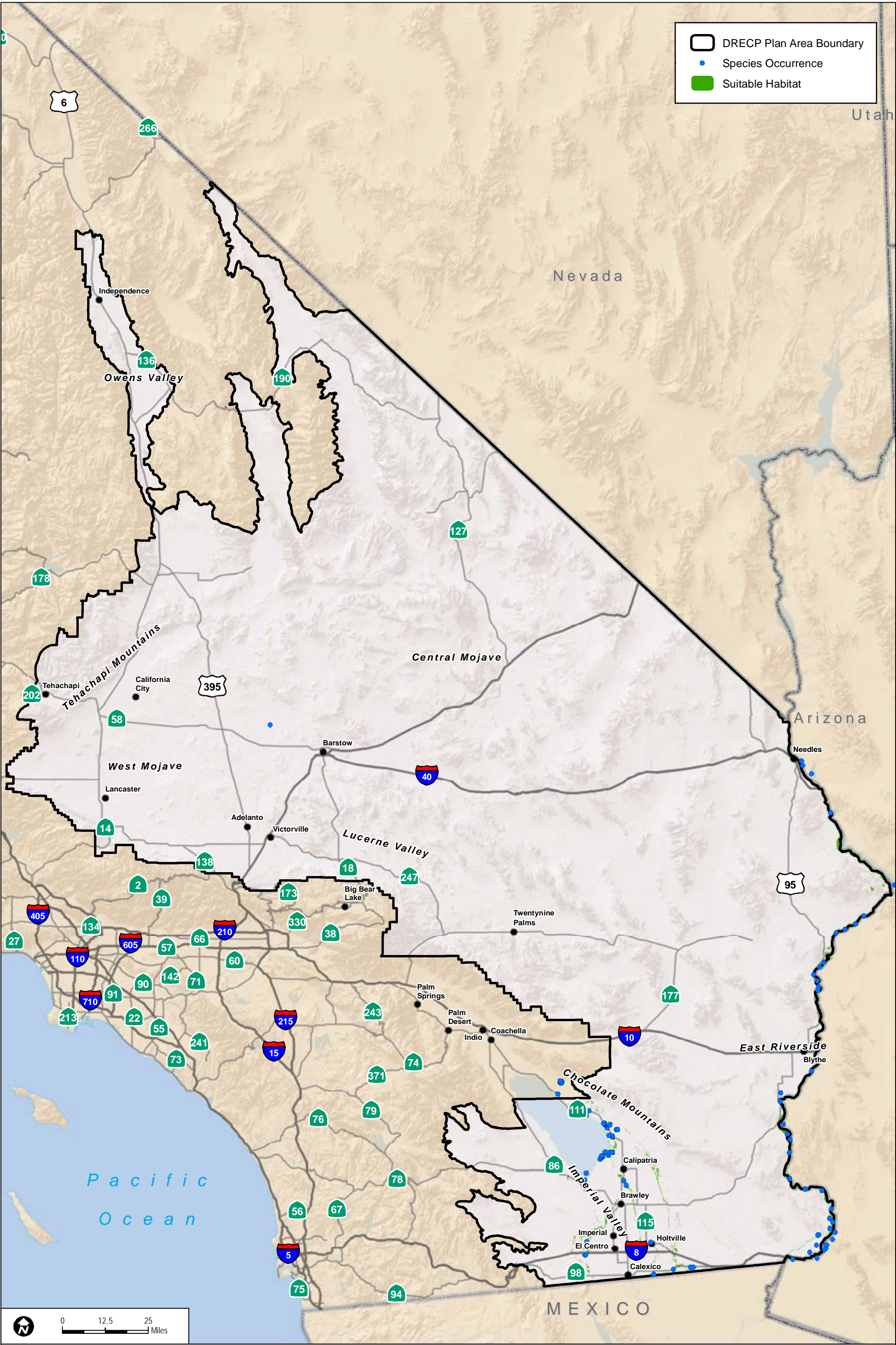
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B12
Draft Species Habitat Model Results for Tricolored Blackbird



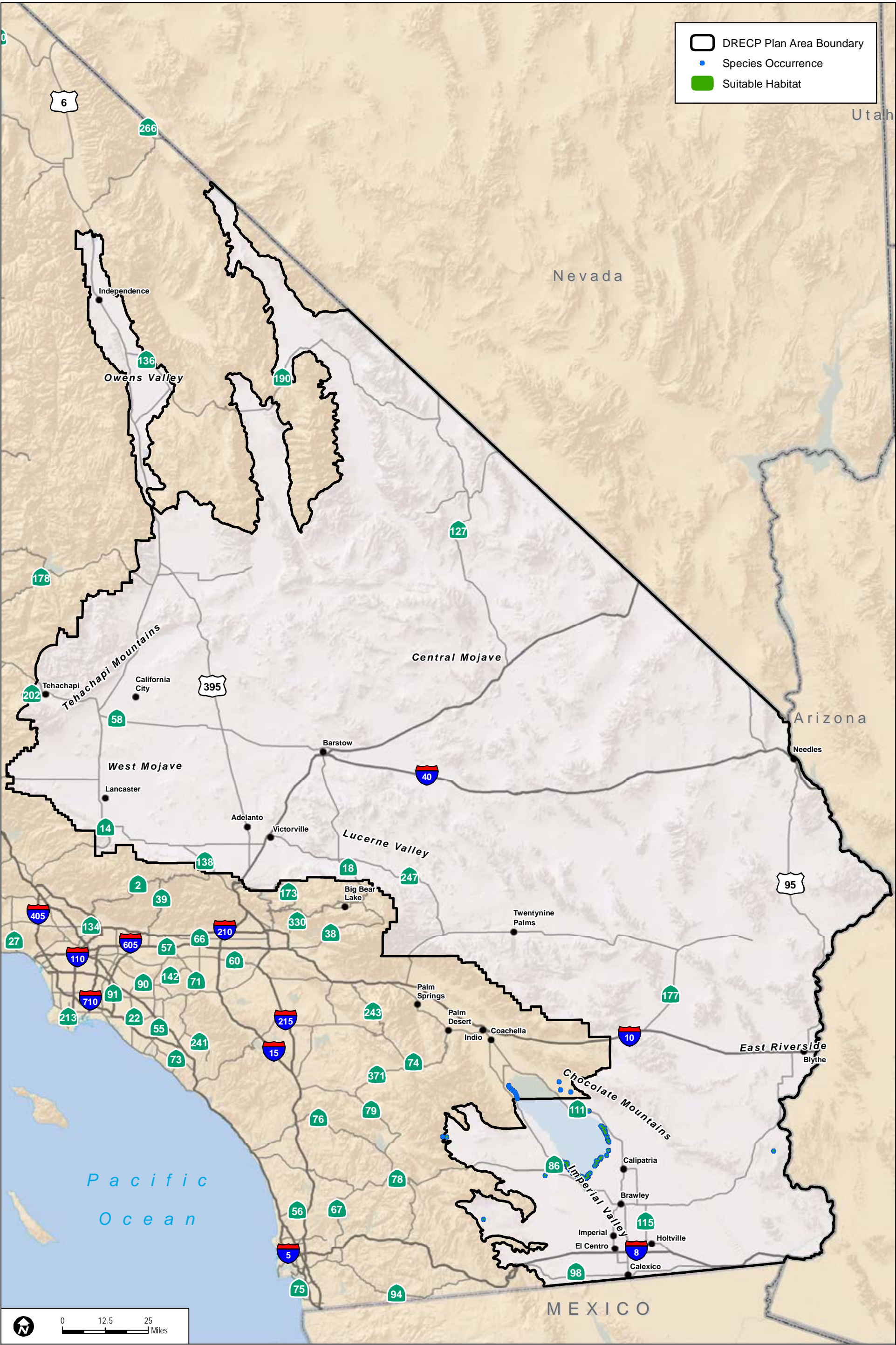
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B13
Draft Species Habitat Model Results for Western Yellow-billed Cuckoo



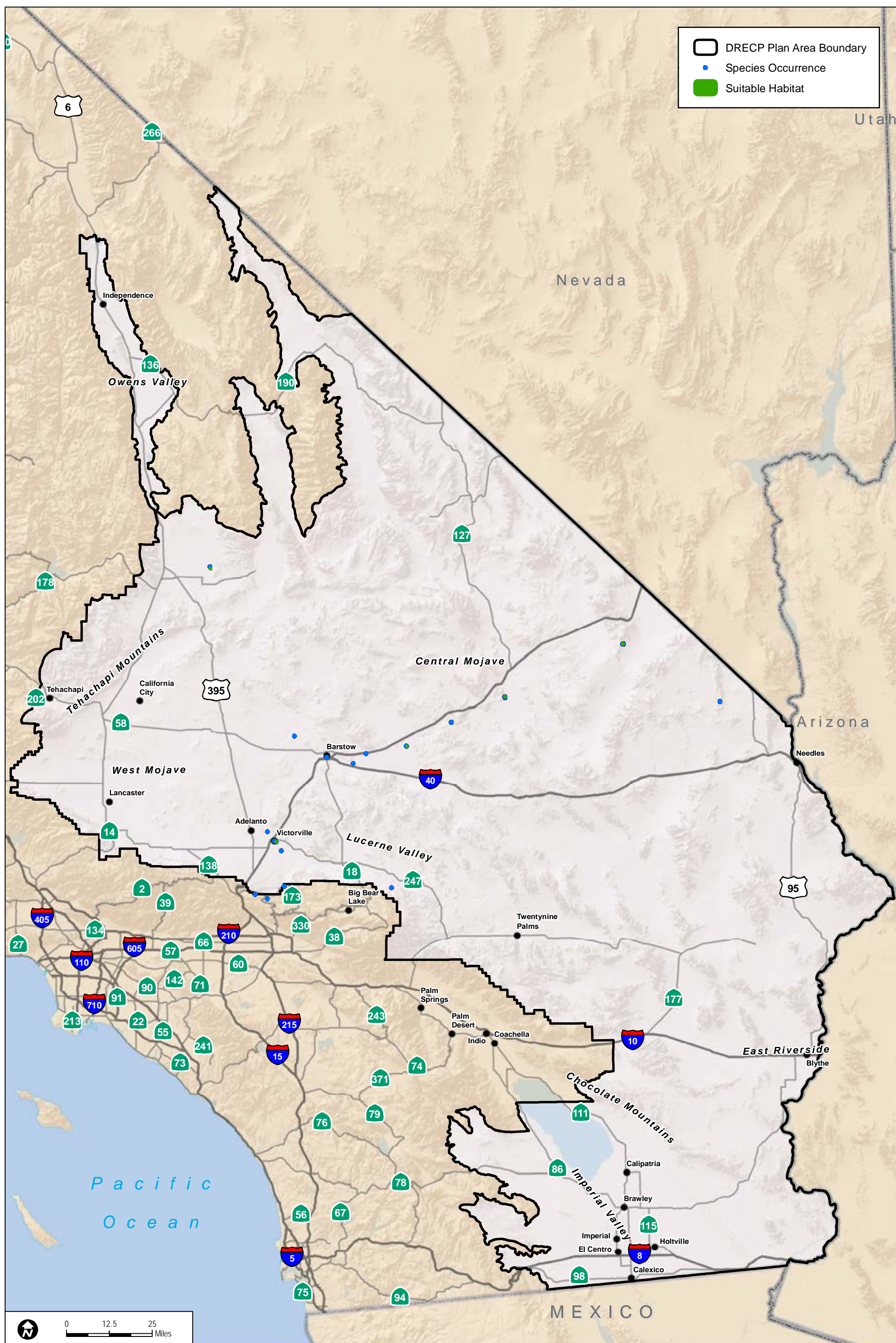
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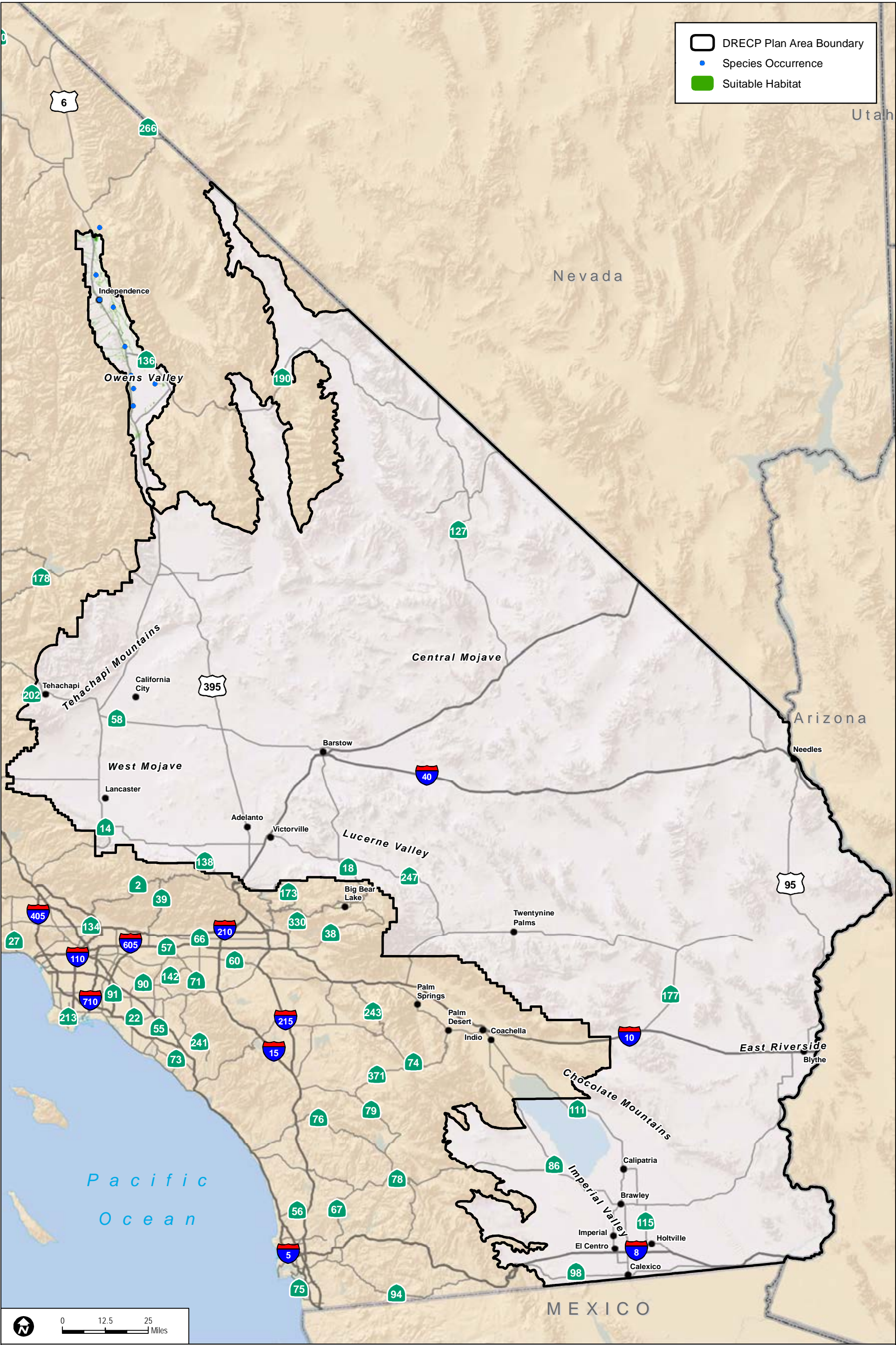
FIGURE SM-B14
Draft Species Habitat Model Results for Yuma Ridgway's Rail



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

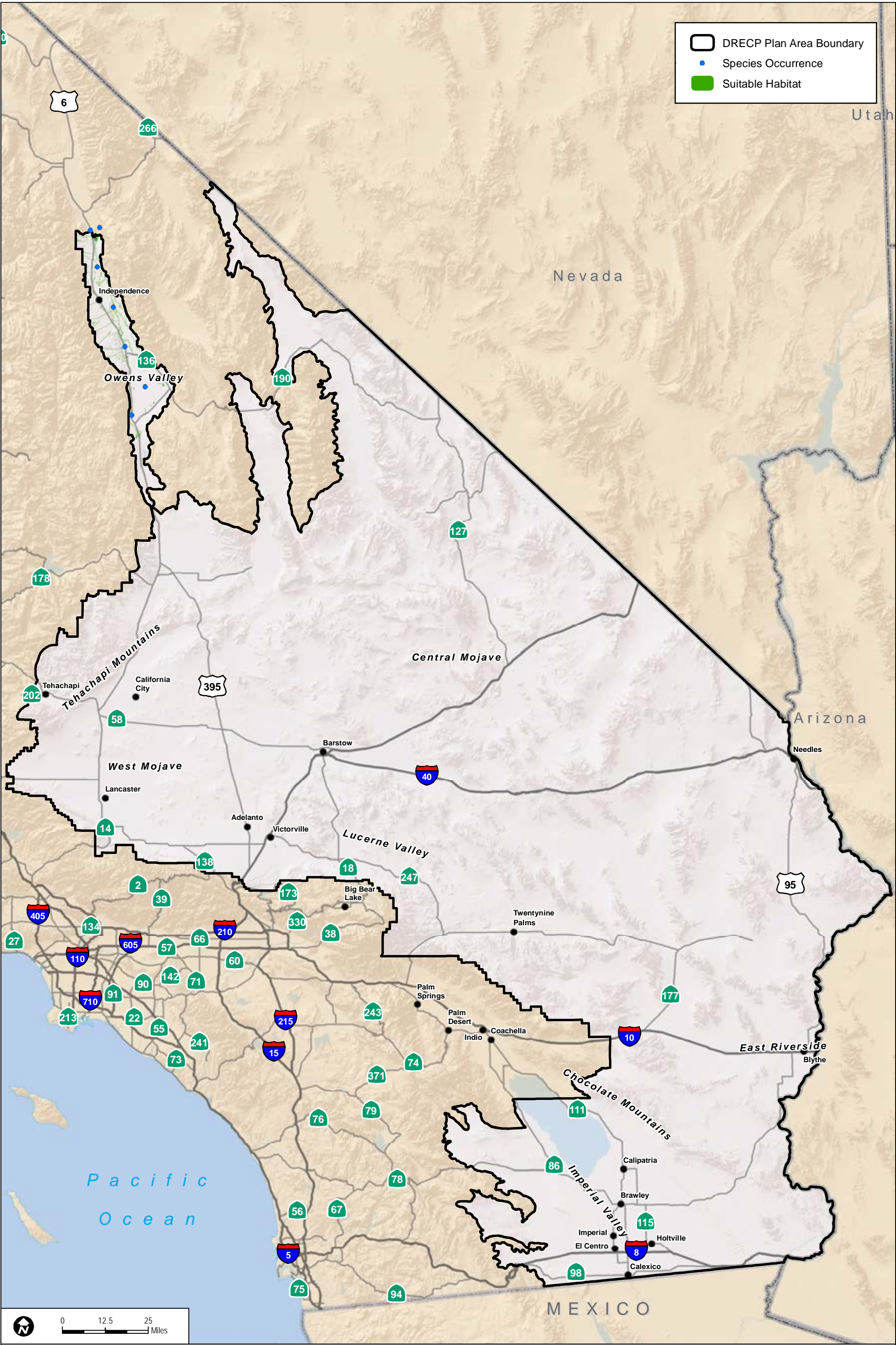
FIGURE SM-F01
Draft Species Habitat Model Results for Desert Pupfish





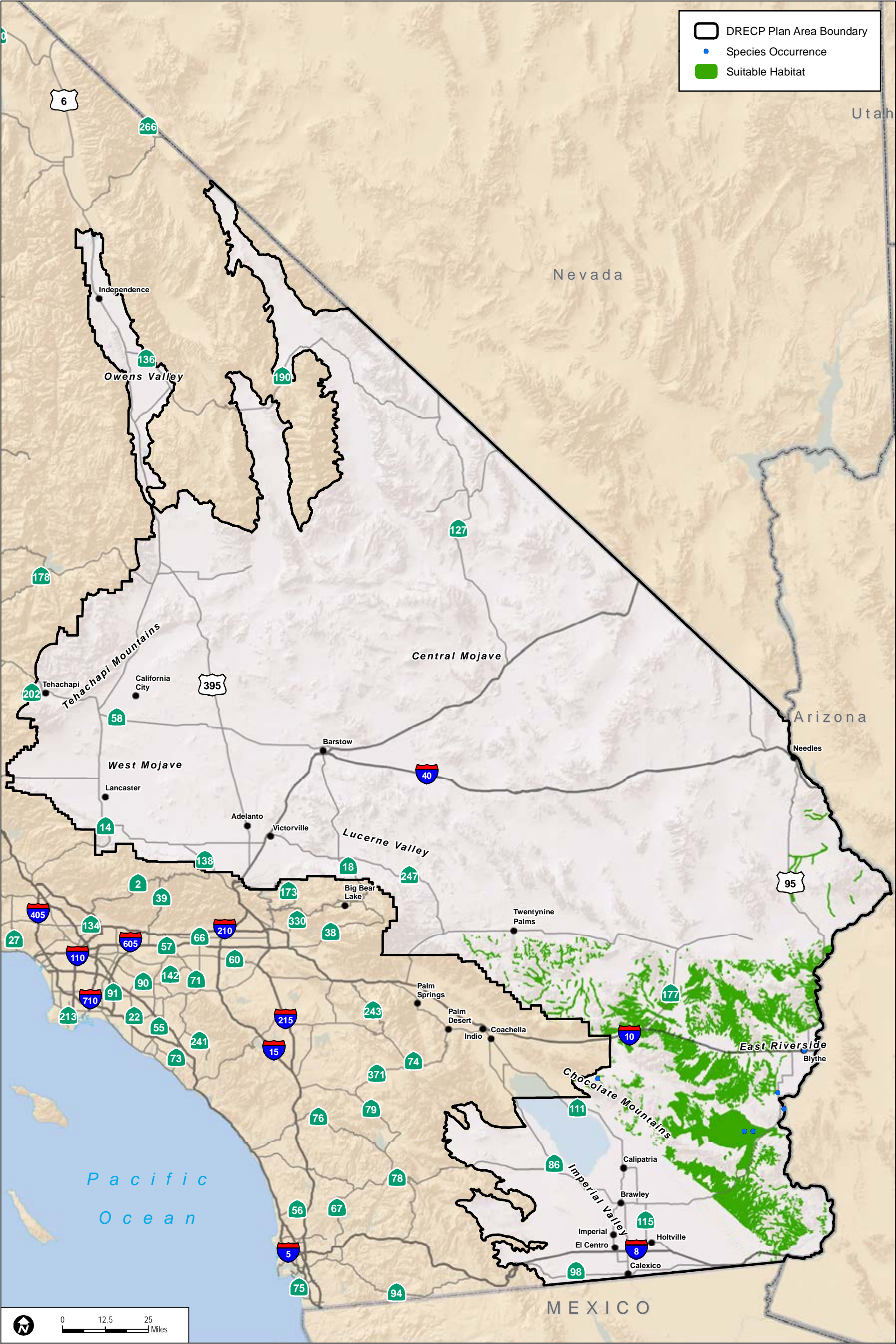
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-F03
Draft Species Habitat Model Results for Owen's Pupfish



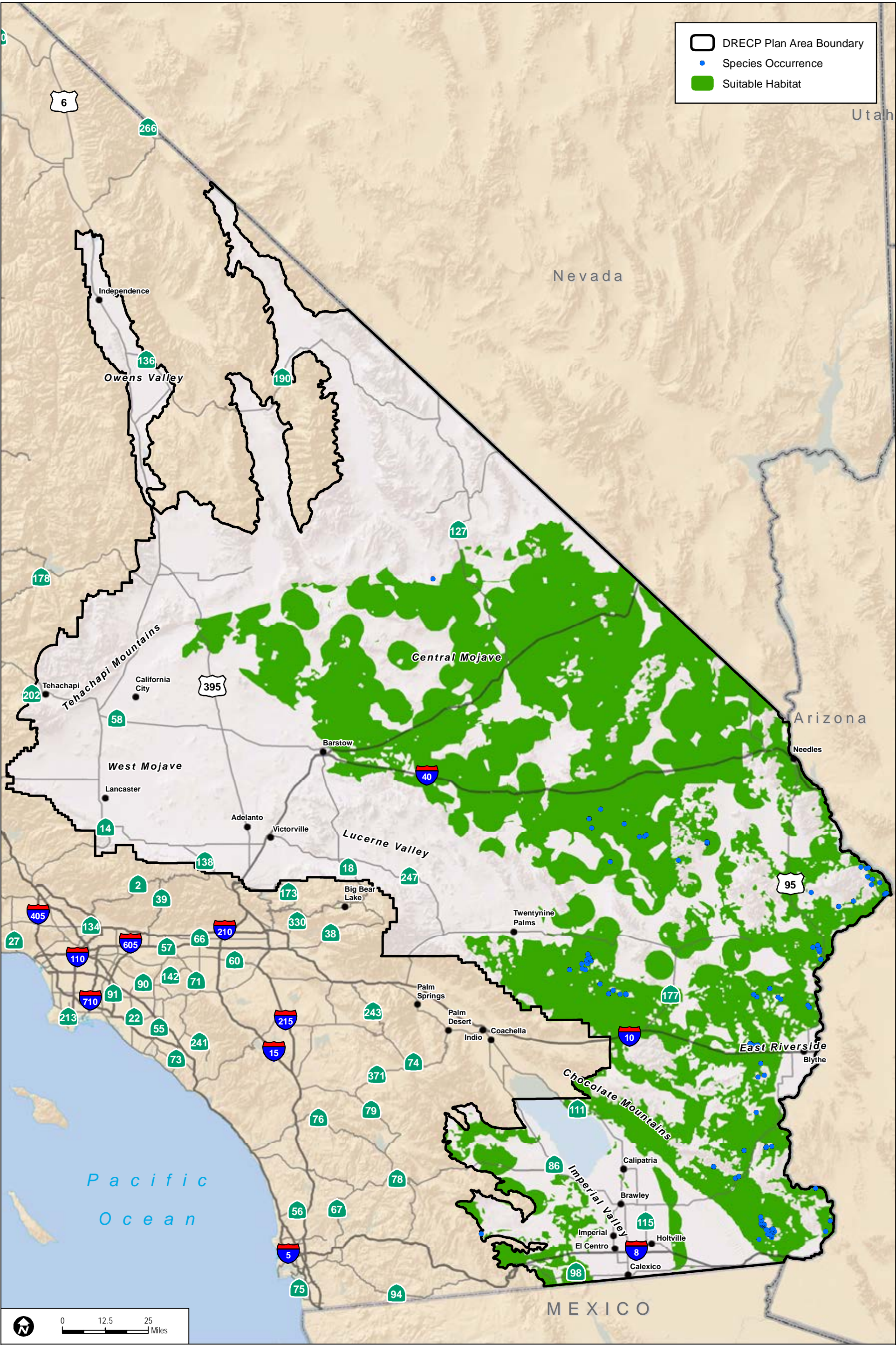
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FIGURE SM-F04
Draft Species Habitat Model Results for Owen's Tui Chub



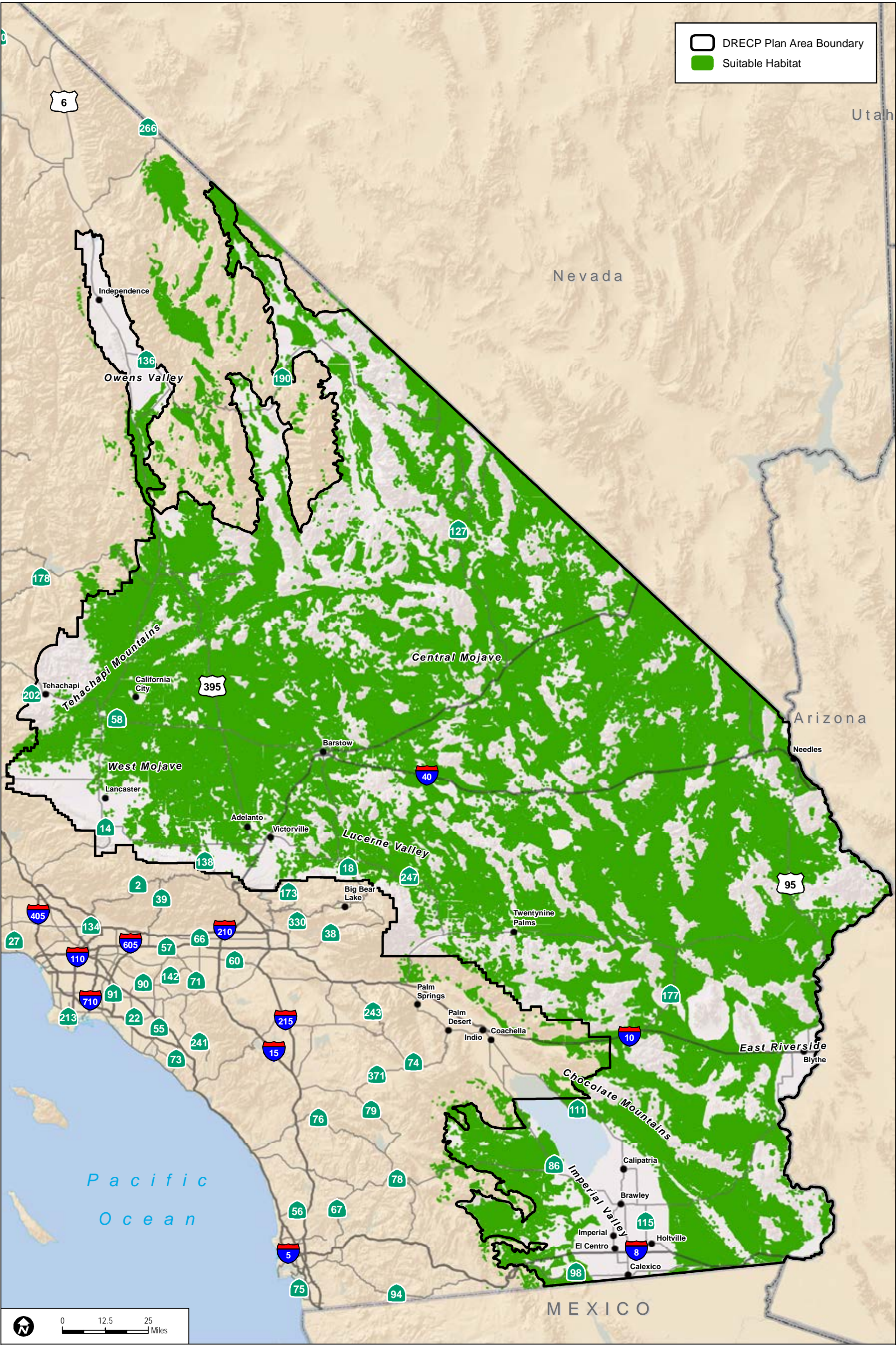
Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M02
Draft Species Habitat Model Results for Burro Deer



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M03
Draft Species Habitat Model Results for Leaf-nosed Bat



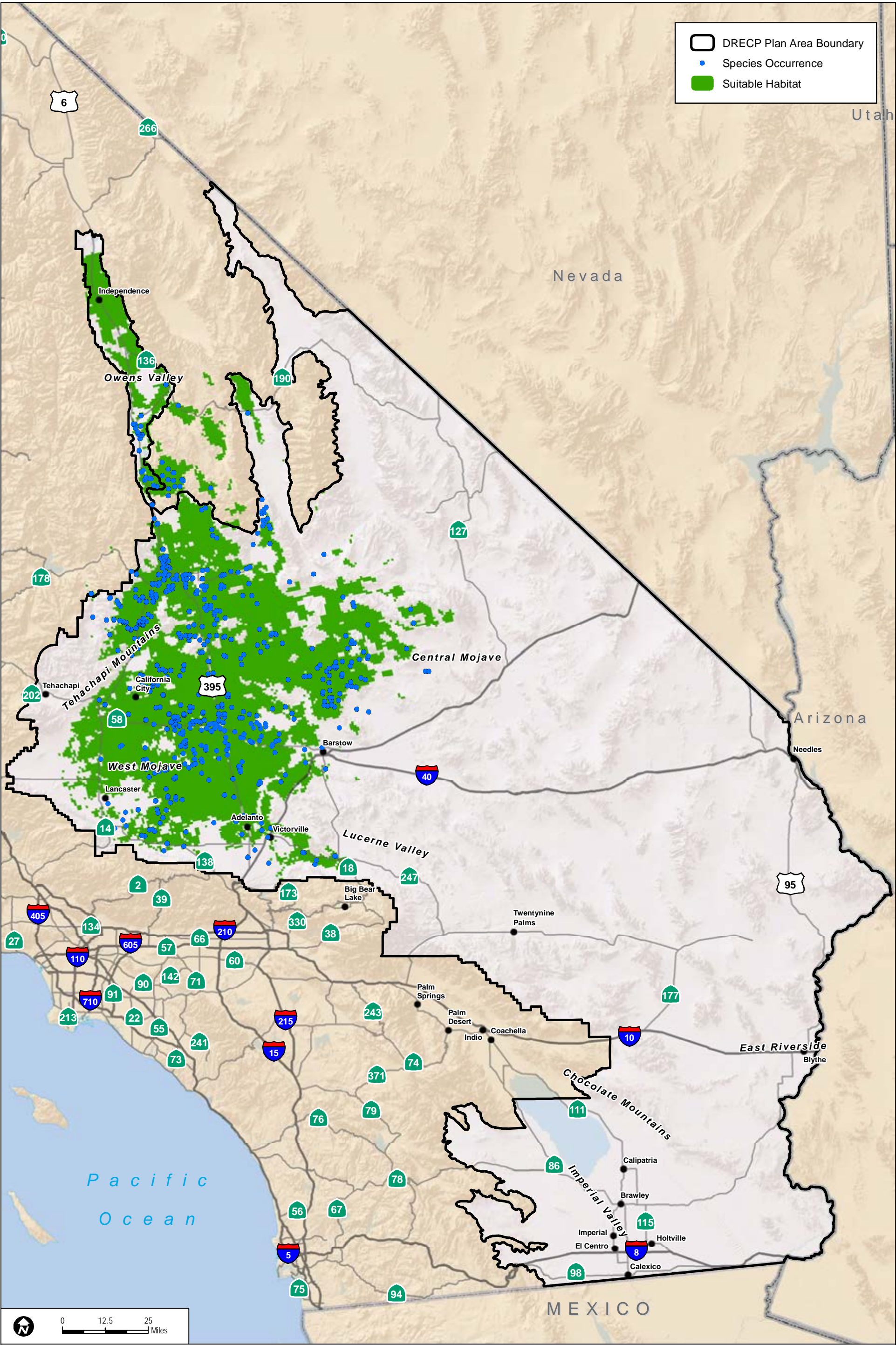
Sources: ESRI (2014); CBI (2013)

FIGURE SM-M04

Draft Species Habitat Model Results for Desert Kit Fox

Desert Renewable Energy Conservation Plan (DRECP) Baseline Biology Report

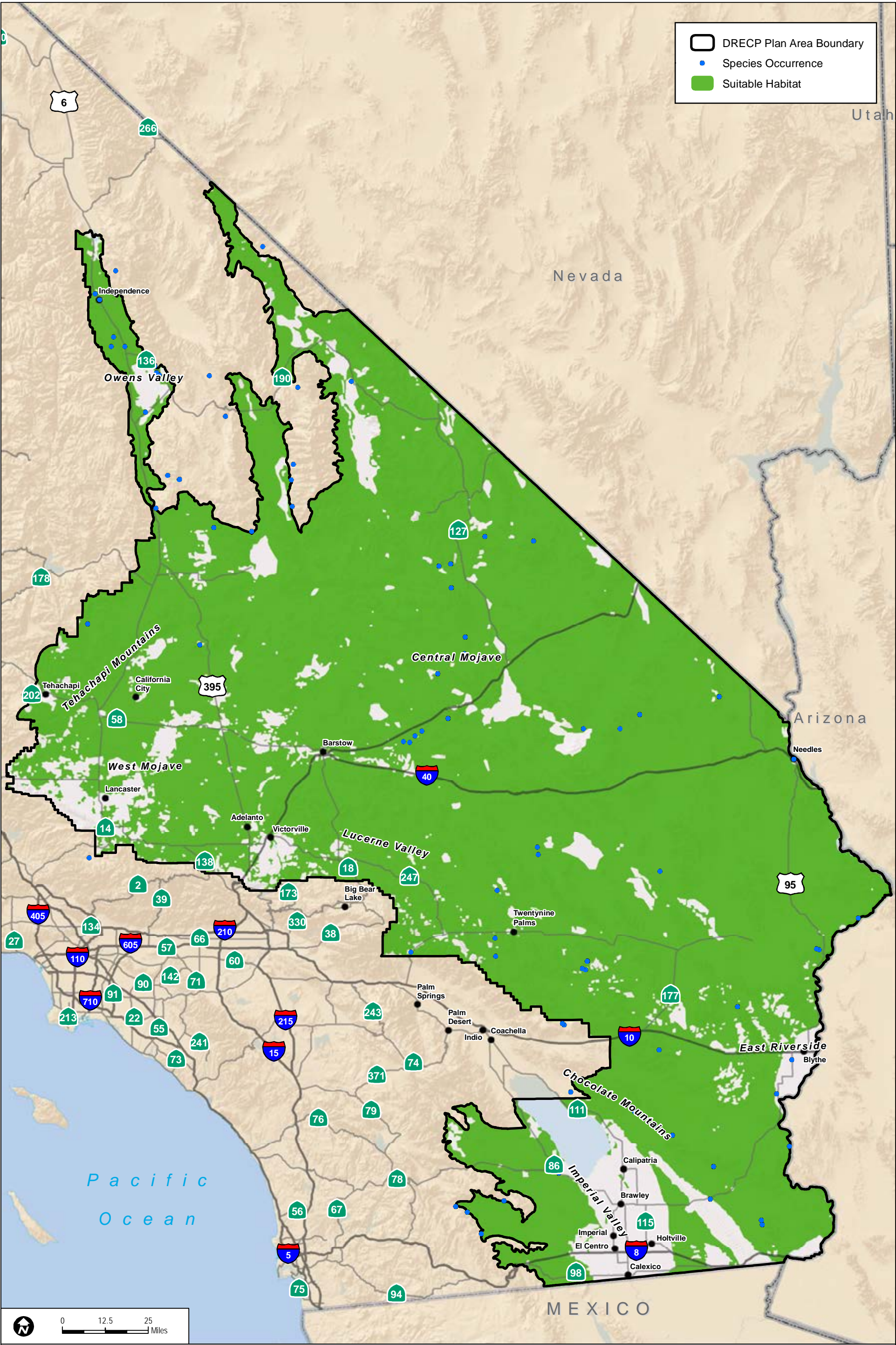
October 2015



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

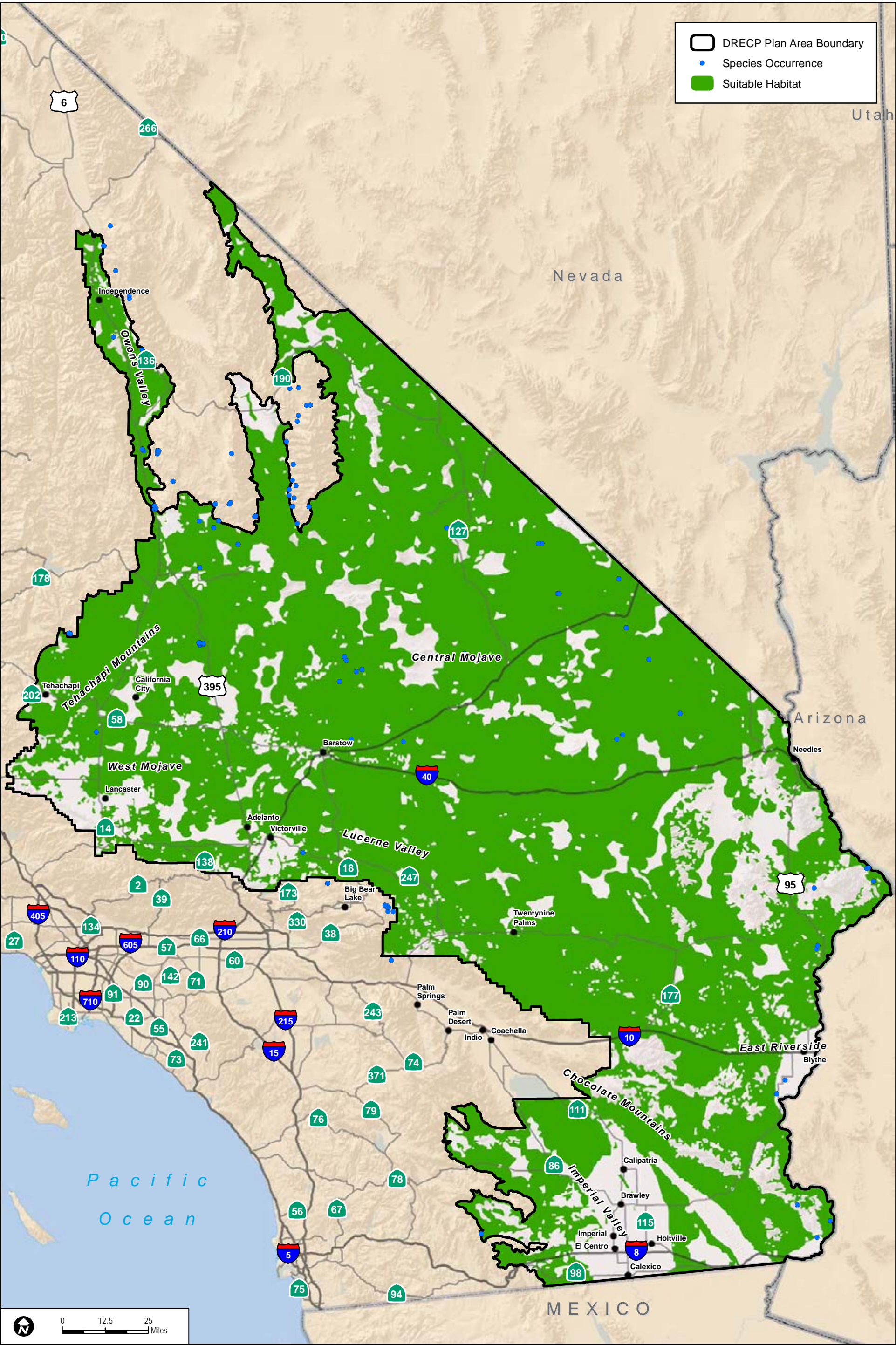
FIGURE SM-M05

Draft Species Habitat Model Results for Mohave Ground Squirrel



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

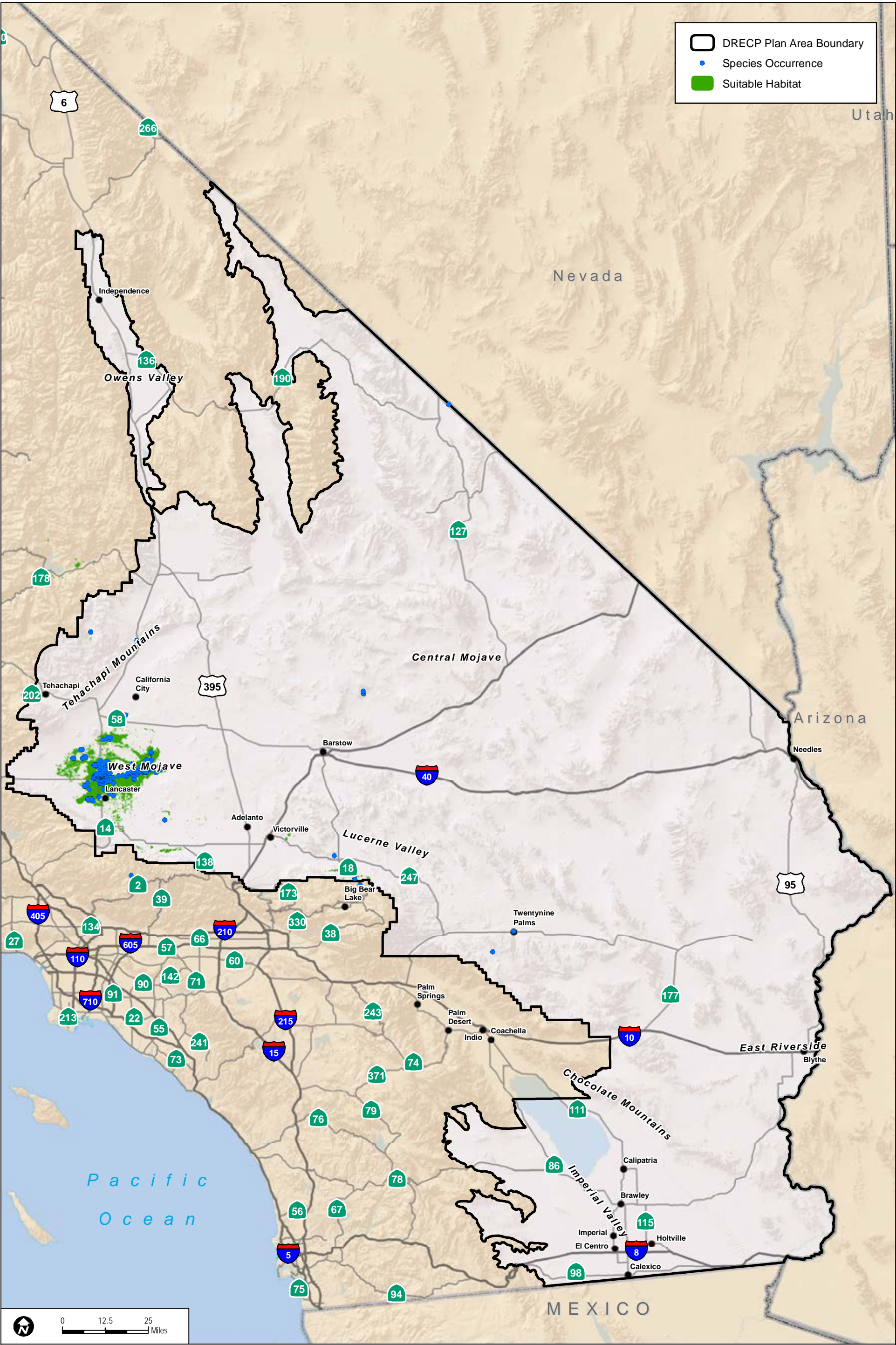
FIGURE SM-M06
Draft Species Habitat Model Results for Pallid Bat



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

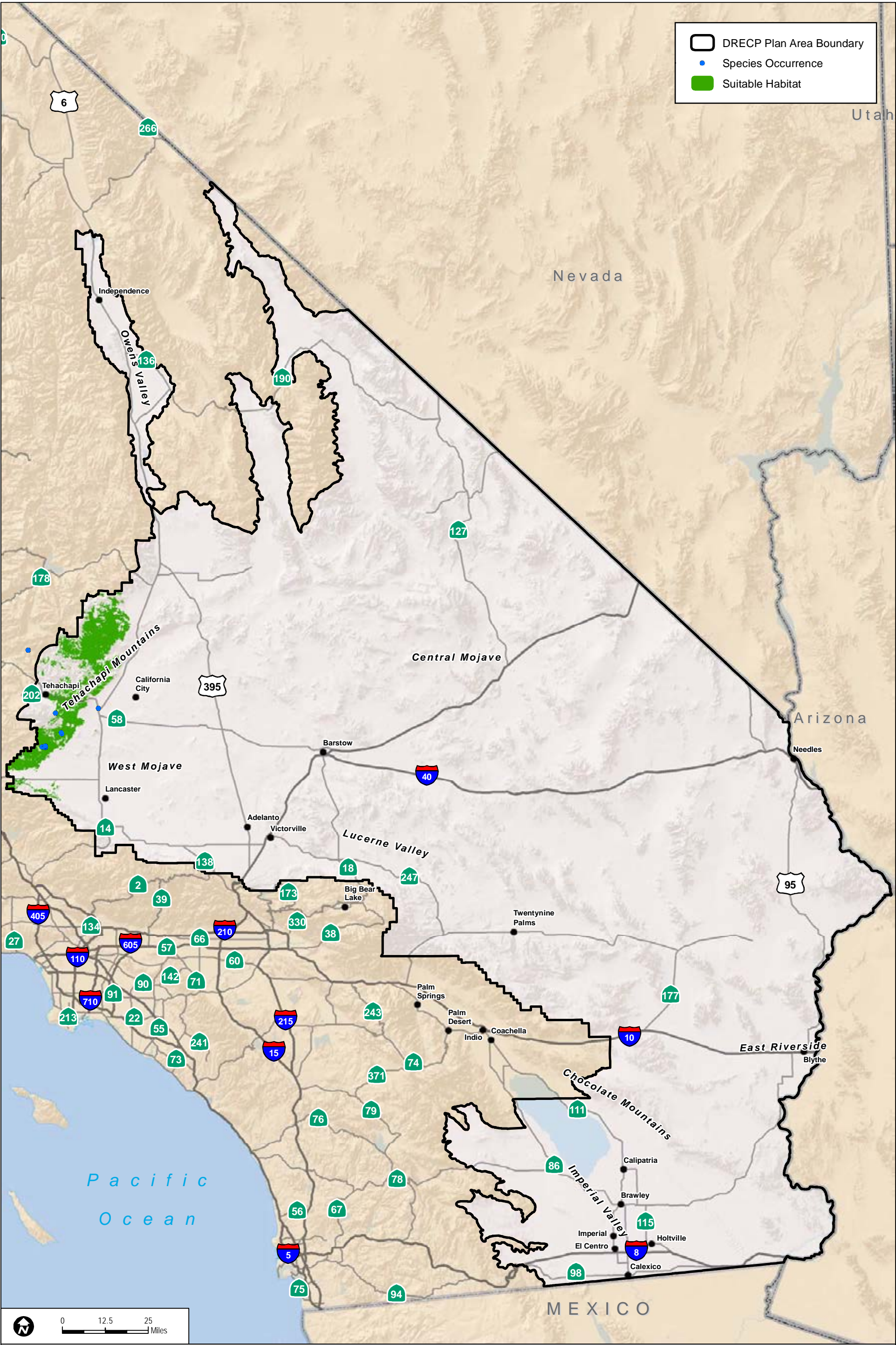
FIGURE SM-M07

Draft Species Habitat Model Results for Townsends Big-eared Bat



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

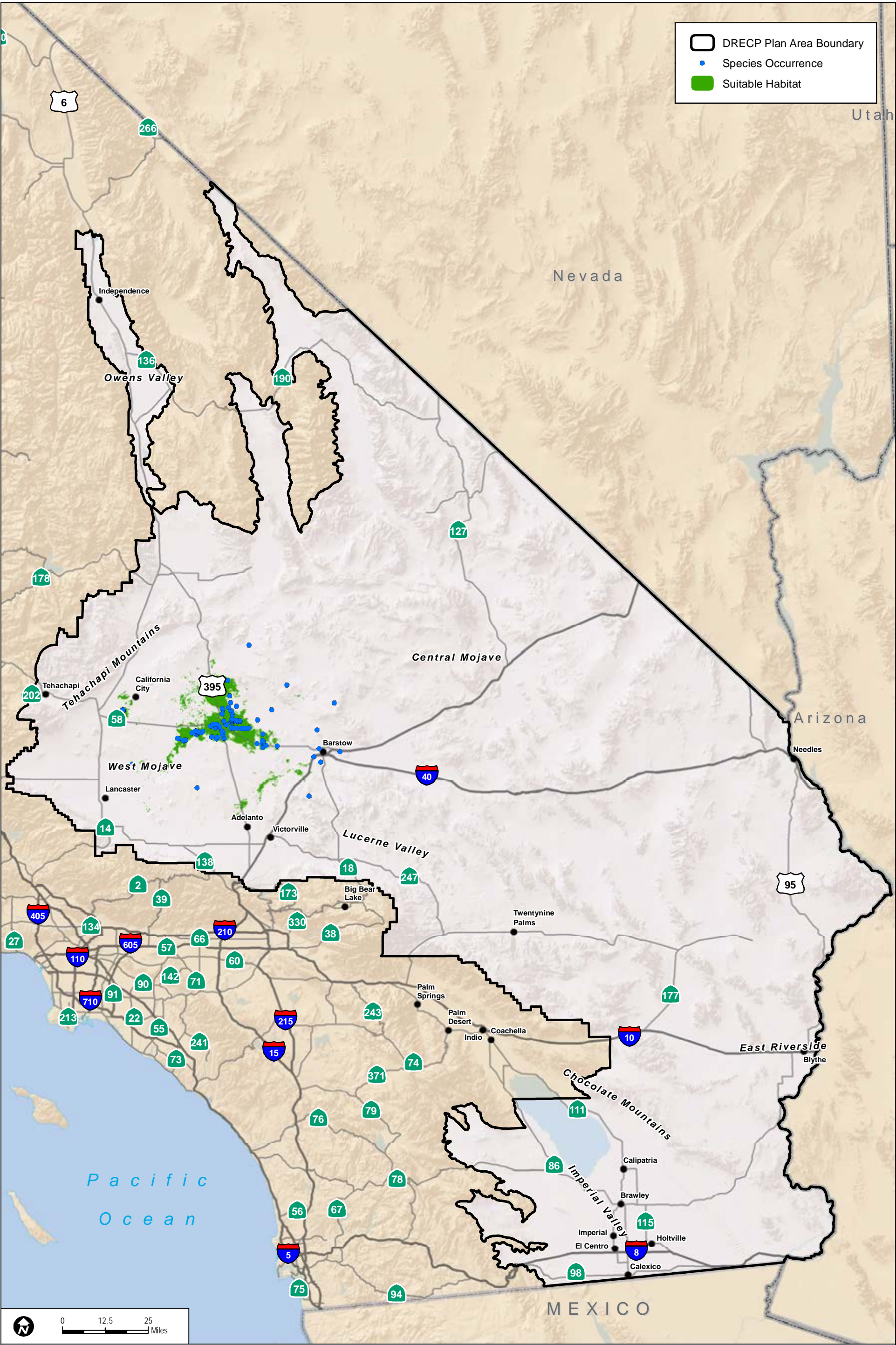
FIGURE SM-P01
Draft Species Habitat Model Results for Alkali Mariposa Lily



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P02

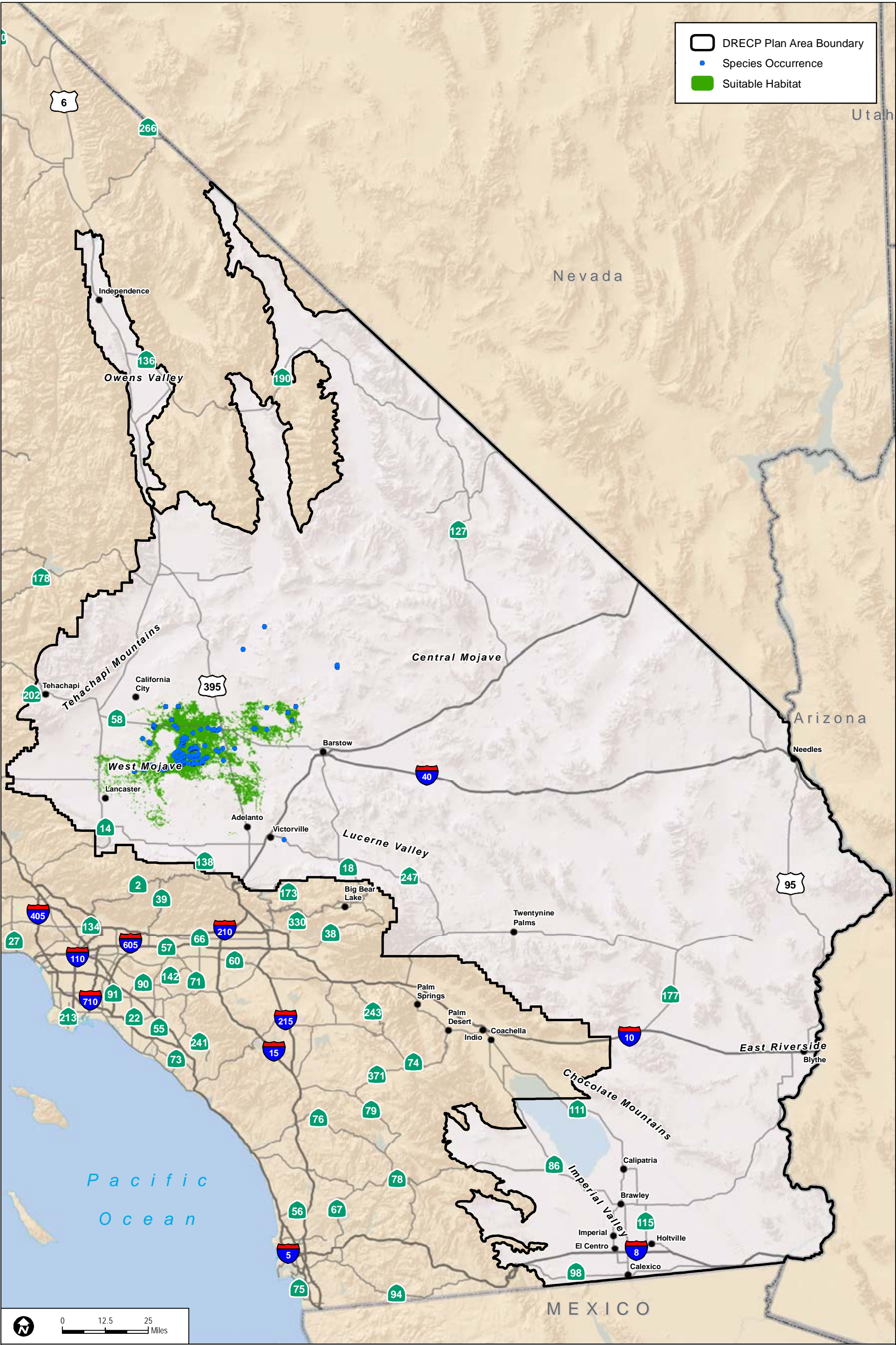
Draft Species Habitat Model Results for Bakersfield Cactus



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

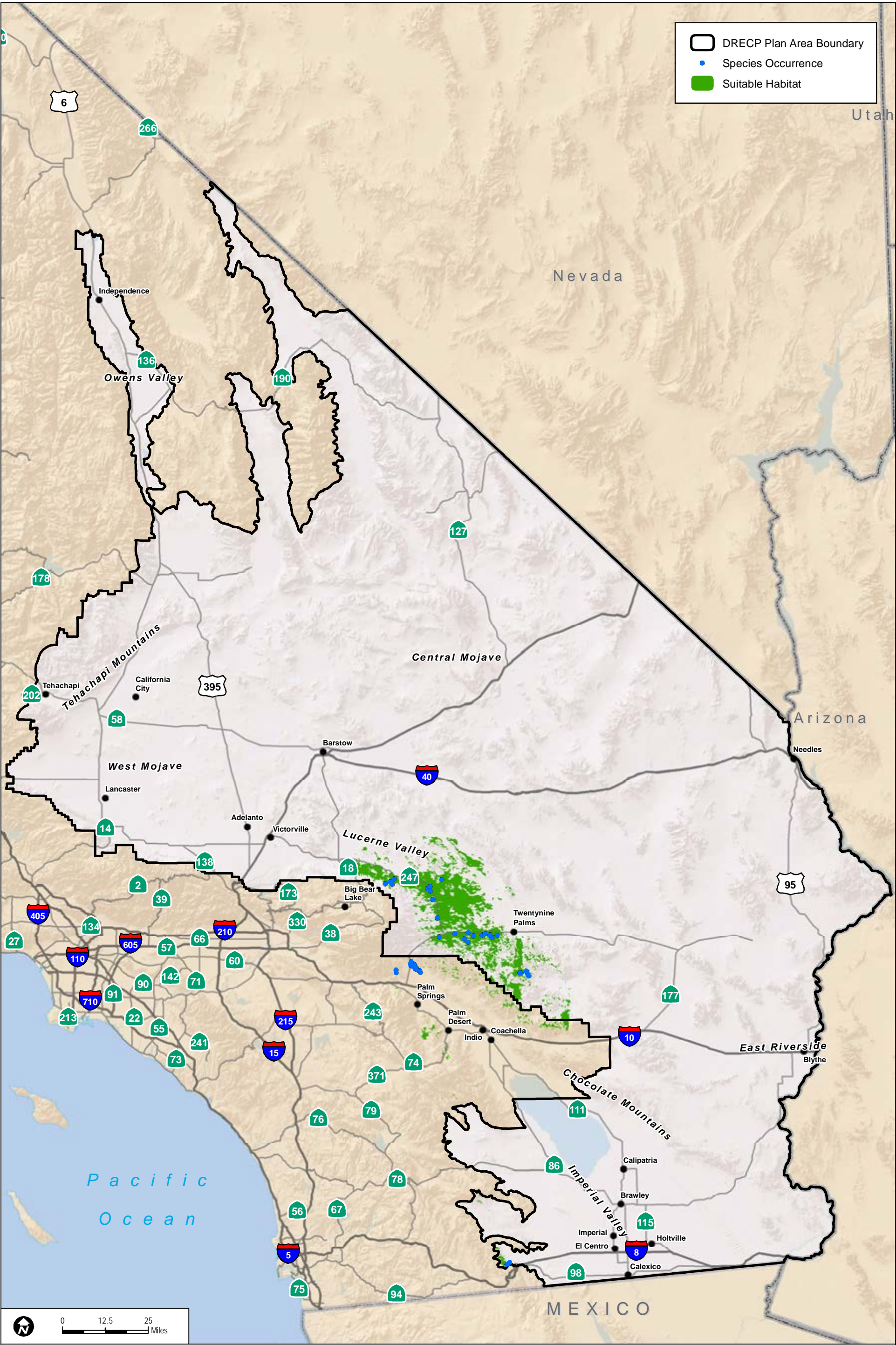
FIGURE SM-P03

Draft Species Habitat Model Results for Barstow Woolly Sunflower



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P04
Draft Species Habitat Model Results for Desert Cymopterus

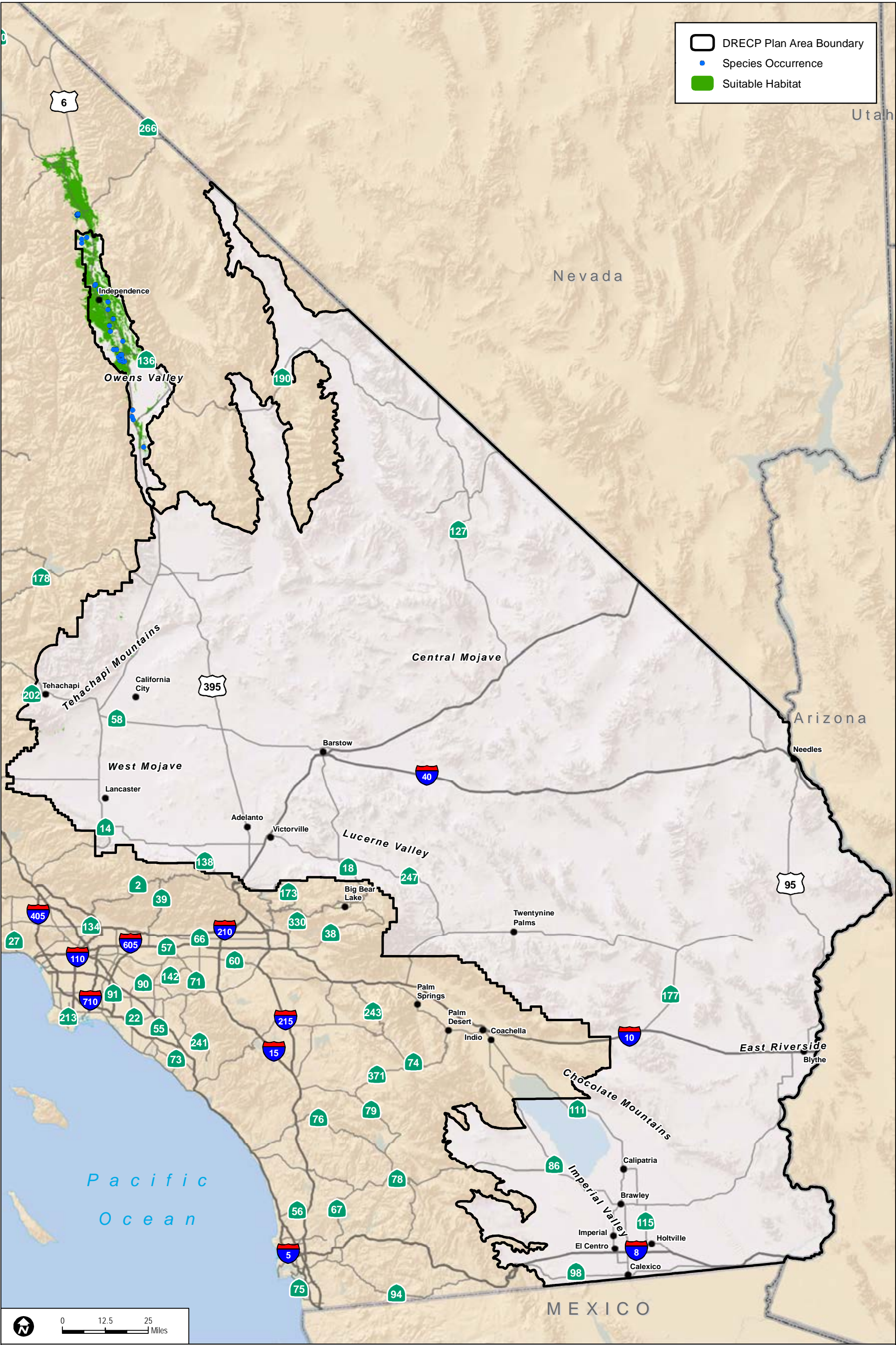


Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P05

Draft Species Habitat Model Results for Little San Bernardino Mountains Linanthus

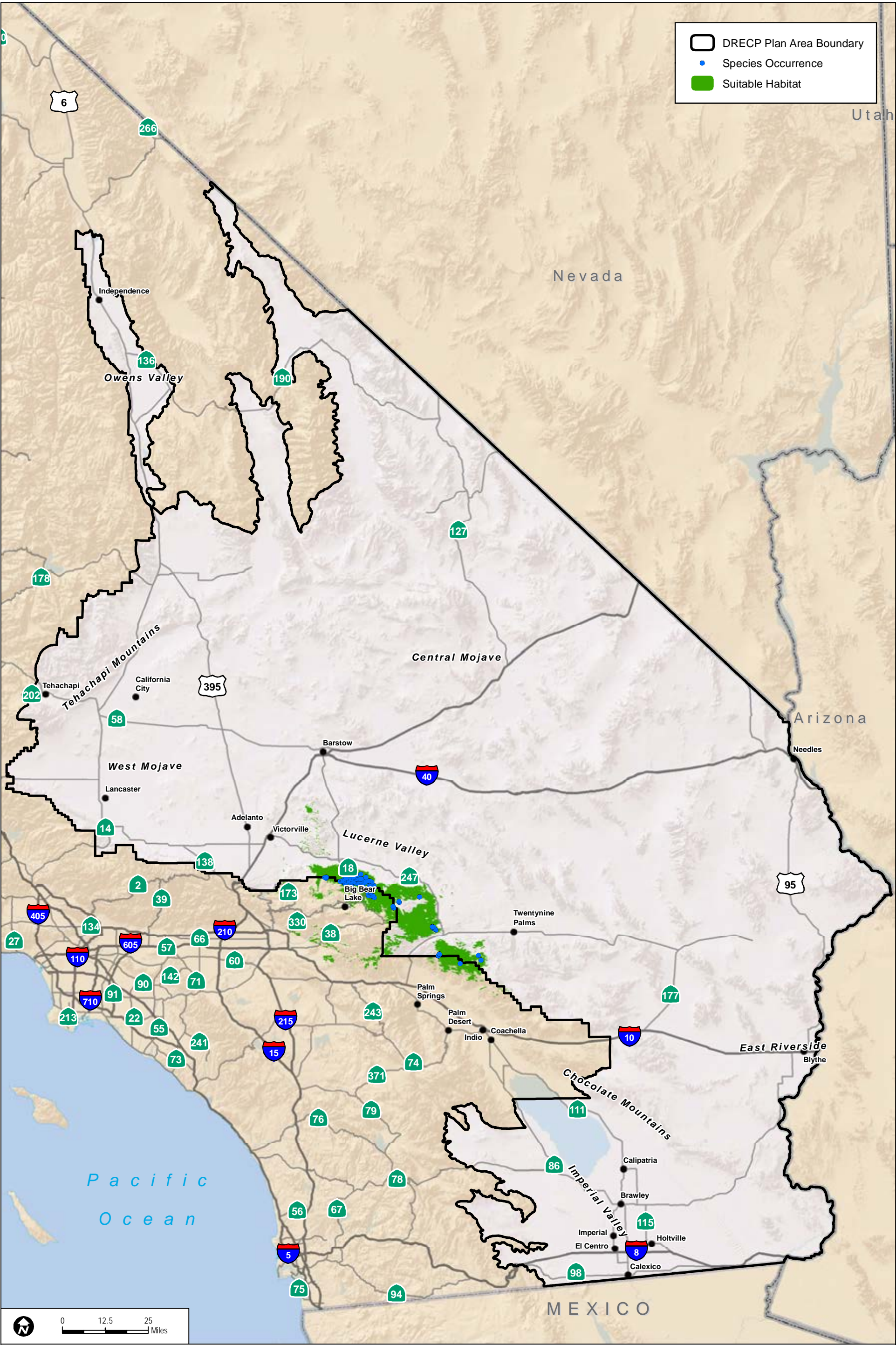




Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

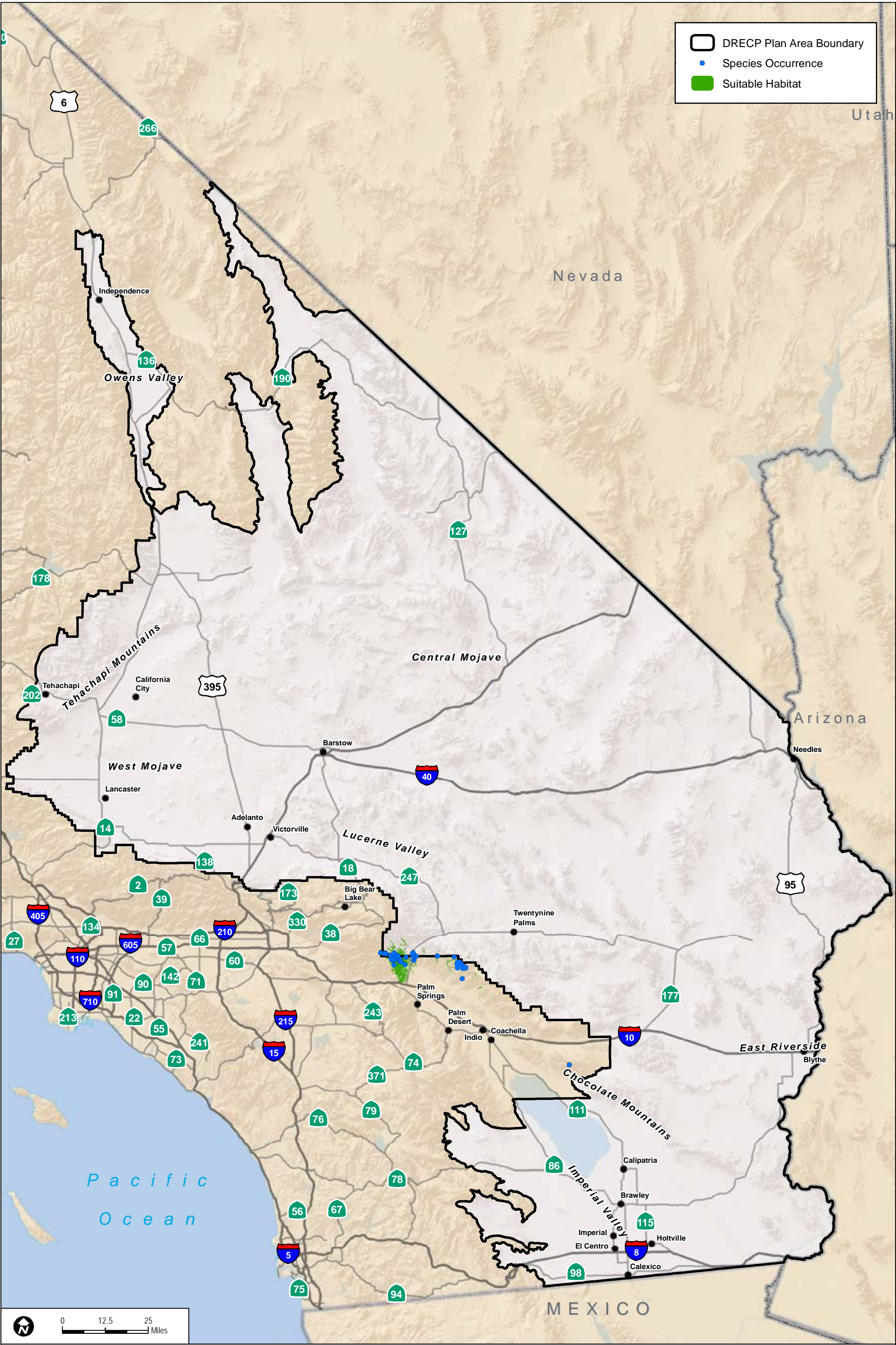
FIGURE SM-P08

Draft Species Habitat Model Results for Owen's Valley Checkerbloom



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

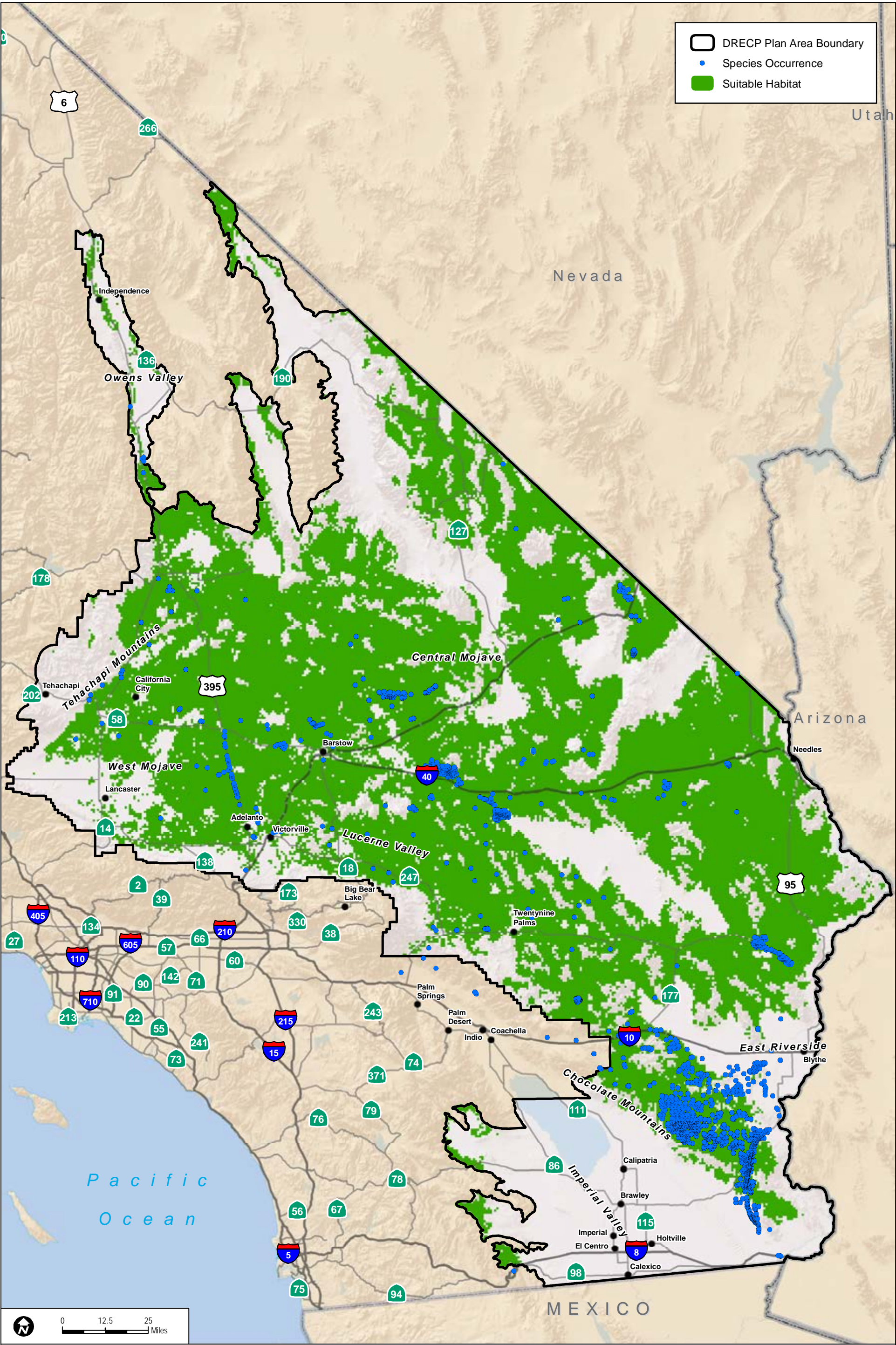
FIGURE SM-P09
Draft Species Habitat Model Results for Parish's Daisy



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

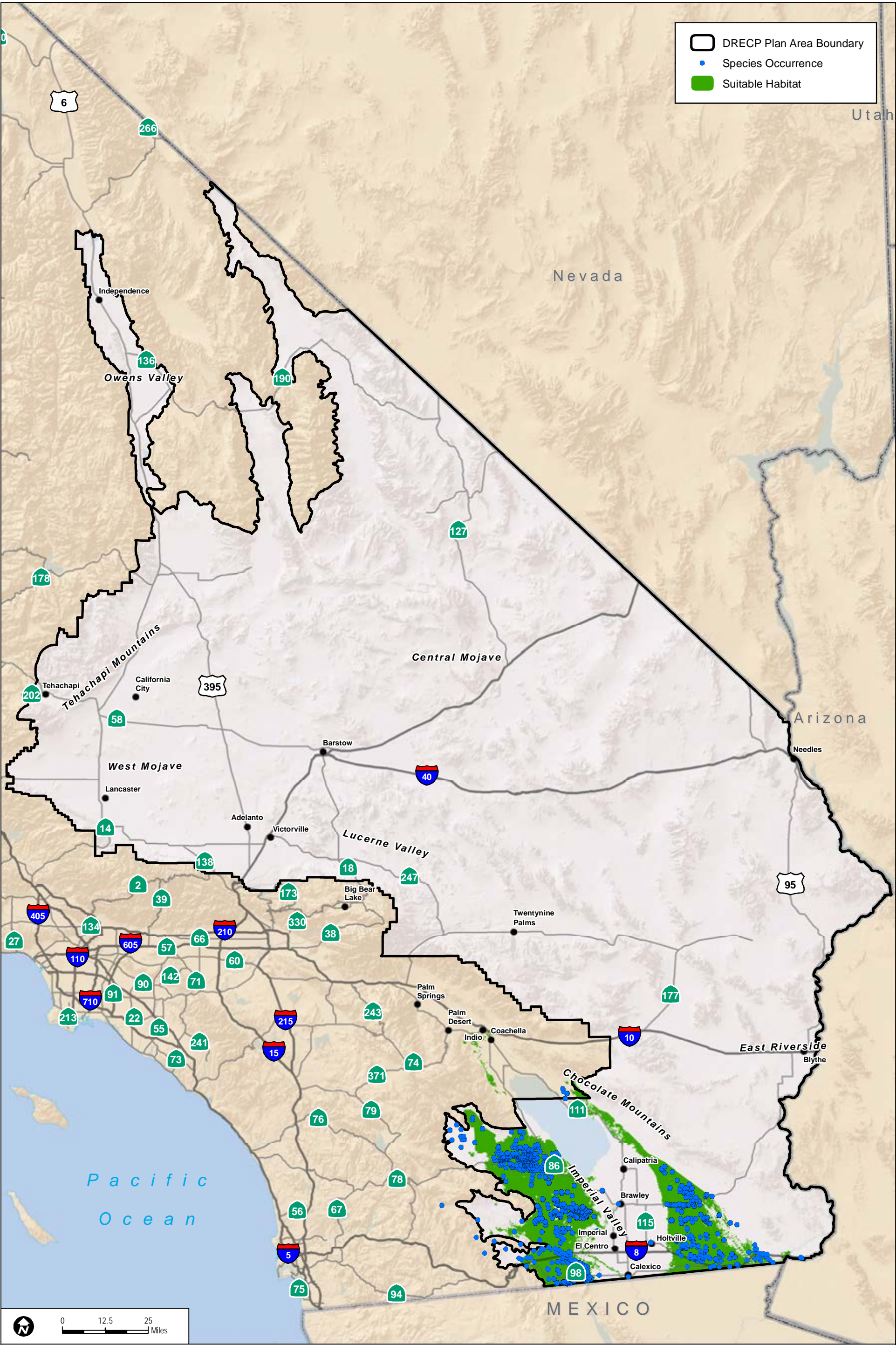
FIGURE SM-P10

Draft Species Habitat Model Results for Triple-ribbed Milk-vetch



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-R01
Draft Species Habitat Model Results for Desert Tortoise



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-R02

Draft Species Habitat Model Results for Flat-tailed Horned Lizard

